Economic and Environmental Input-Output Modeling: Building Material Recycling

A Dissertation
Presented to
The Academic Faculty

by

Taelim Choi

In Partial Fulfillment
of the Requirements for the Degree
Doctor of Philosophy in the
School of City and Regional Planning

Georgia Institute of Technology
December 2012

Economic and Environmental Input-Output Modeling:
Building Material Recycling

Approved by:

Dr. Nancey Green Leigh, Advisor
School of City and Regional Planning
Georgia Institute of Technology

Dr. Steve P. French
School of City and Regional Planning
Georgia Institute of Technology

Dr. Matthew J. Realff
School of Chemical & Biomolecular Engineering
Georgia Institute of Technology

Dr. Randall W. Jackson
Department of Geology and Geography
West Virginia University

Dr. Subhro Guhathakurta
School of City and Regional Planning
Georgia Institute of Technology

Date Approved: October 23, 2012

ACKNOWLEDGEMENTS

I would like to thank Dr. Nancey Green Leigh, my advisor and mentor. She provided invaluable opportunities and financial support for my entire doctoral studies. Thanks to her support, guidance, advice, patience, and encouragement, I could complete the dissertation. I also wish to thank Dr. Randall W. Jackson of the Regional Research Institute at West Virginia University. My research benefited greatly from his advice and expertise. I would also like to recognize my three committee members, Dr. Steve P. French, Dr. Subhro Guhathakurta, and Dr. Matthew J. Realff for their helpful comments and insights on the dissertation. I also wish to thank my minor advisor Dr. Douglas Noonan in the School of Public Policy. Discussions and readings from his classes became valuable inputs for my research.

I am especially indebted to the Sustainable Industrial Systems for Urban Regions (SISFUR) research team for allowing me to connect with excellent research fellows. I also wish to thank Jane Chisholm for her efforts in editing a draft of this dissertation. Thanks to my friends and colleagues in the School of City and Regional Planning at Georgia Tech. Of course, none of this would have been possible without the prayers and encouragement of my mother and father. Finally, I am truly thankful to my wife and best friend, Yeonggyeong Jee for her love, support, patience, and dedication. You are always by my side. Thank God with all my heart. The Lord is my shepherd, I shall not be in want.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Page

SUMMARY

A key dimension to improving urban economic and environmental sustainability is the efficient use of resources through recycling. A thriving recycling system requires not only effective institutional policies and community-wide diversion efforts, but also a competent local and regional recycling industry. Although the recycling industry has traditionally been recognized as a local service and fringe industry, it has noticeably transformed into an integral segment of industrial production systems as manufacturers have increasingly begun to adopt the principle of extended producer responsibility. Despite such changes, urban and regional theory and planning research has largely disregarded the industrial aspect of recycling, contributing to the dearth of information about the organizational and spatial patterns of the recycling industry and the impact of the establishment of recycling systems on local and regional scales.

Given the knowledge gap, this dissertation addresses two questions: 1) What is the logic of the industry organization and spatial pattern of recycling industry in different institutional contexts? and 2) How is the economic and environmental impact of recycling systems determined in cases of construction and demolition waste recycling and waste carpet recycling? To answer the first question, this research develops a theoretical model that explains how recycling industrial activities are spatially distributed in light of institutional and organizational theories. The theoretical model characterizes organizational decisions pertaining to recycling functions and suggests spatial patterns of recycling systems.

With respect to the second question, this research constructs a regional environmental input-output model on the metropolitan scale. It estimates regionalized energy use coefficients and greenhouse gas emission coefficients using various sources of data mainly compiled from the Manufacturing Energy Consumption Survey 2006, the State Energy Consumption Estimates, and the Commodity Flow Survey 2007. Based on regional input-output tables coupled with the regionalized environmental coefficients, this research quantifies, through simulations, the net economic and environmental impact of a localized construction and demolition waste recycling system in the San Francisco metropolitan area and regional carpet recycling systems in the Atlanta and Seattle metropolitan areas.

Results of the simulations reveal that 1) the localized construction and demolition waste recycling system provides moderate economic benefits because of the limited job creation potential of mechanized recycling processes and yields relatively small environmental benefits with respect to the total weight processed; 2) wider adoption of the deconstruction technique expands job opportunities, increases energy savings, and reduces greenhouse gas emissions during the course of construction and demolition waste recycling; 3) regional-scale waste carpet recycling systems, in particular recycled nylon 6 production, create sizable new job opportunities and provides environmental benefits of energy savings and greenhouse gas emission reduction despite the long-distance transportation of waste carpet. These results suggest that policies that promote recycling industrial activities can significantly contribute to the economic and environmental sustainability of metropolitan areas.

CHAPTER 1. INTRODUCTION

The integration of local economic development and environmental sustainability has been the focus of increased attention in planning research and practice. Fitzgerald and Leigh (2002) proposed the principle of sustainable local economic development, the objectives of which are to increase the standard of living, to reduce inequality, and to promote sustainable resource use and production. Despite recognizing the call for balanced local economic development planning, only a few pioneering planning researchers have attempted to address specific strategies that foster sustainable local economic development (Leigh and Patterson, 2006; Fitzgerald, 2010). Furthermore, local economic development planners have not widely acknowledged the importance of sustainability as a practical strategy (Patterson, 2007; Jepson, 2004; Saha and Paterson, 2008).

The dearth of planning research on sustainable local economic development may be associated with the following reasons. First, the importance of physical materials and energy resources has dramatically declined in the theories of urban and regional growth. In classical location theory, material input resources associated with transportation costs were one of the key factors explaining the spatial patterns of industry. However, in the global economy, as the well-established transportation infrastructure has significantly lowered transportation costs, the significance of material resources as a location factor has weakened. The management of material resources was not considered to be a factor strongly influencing economic growth and development in traditional urban and regional theories. Consequently, the topics on the physical side of the industrial system, particularly those pertaining to sustainable production and efficient resource use, have been largely disregarded in research pertaining to urban and regional growth. Only a few recent inter-disciplinary studies such as the theory of ecological modernization and eco-industrial park development have examined efficient resource use, cleaner production, and waste exchange in the context of local economic development (Gibbs, 2006; Deutz and Gibbs, 2004).

In addition to declining interest in physical input resources, the recycling system and the spatial pattern of recycling-related facilities have been scarcely illuminated in the light of urban and regional growth theories. Recycling was conventionally understood as a local service sector and regarded a fringe industrial sector, but such recognition scarcely acknowledges current changes in industry. Recycling industrial activities have been noticeably transformed in the emergence of the principle of extended producer responsibility (EPR), in which manufacturers should take responsibility for managing their end-of-life (EOL) products (Lifset, 1993; Toffel, 2003). The involvement of manufacturers in recycling has created new dynamics in institutional rules and spatial linkages of recycling systems. Nevertheless, the lack of research interest in the recycling industry has resulted in under-theorization regarding the growth of the recycling industry and its spatial patterns.

Another factor for the scarcity of empirical and practical studies is the lack of sub-national level data on material and energy consumption and waste generation. Since the late 1960s, several researchers (Isard, 1968; Ayres and Kneese, 1969; Leontief, 1970; Johnson and Bennett, 1981; Briassoulis, 1986; Huang et al., 1994) have conceptually discussed and developed the environmental input-output (IO) model, which allows the investigation of economic and environmental interactions. Such models extended the conventional IO framework, which examined how industrial activities influence the local economy and the environment by creating a new industry sector of pollution reduction and connecting economic and ecological spheres. However, lacking relevant region-specific information, many studies have constructed only hypothetical scenarios, and they have not been adopted into planning practice.

Purpose of Research

With respect to the gap in knowledge of the theoretical foundation of the recycling industry and the analytical model of sustainable local economic development, this dissertation addresses two related questions: 1) What is the logic of the industry organization and spatial pattern of the recycling industry in different institutional contexts? And 2) how is the economic and environmental impact of recycling determined in cases of construction and demolition waste (CDW) recycling and waste carpet recycling? This dissertation attempts to answer these questions through the development of theoretical and analytical models.

This research seeks to develop a theoretical model that explains the growth pattern of the recycling industry. Specifically, it examines how the recycling industry is organized and spatially distributed in the different institutional approaches. It proposes the schematic framework that identifies the key decision makers on organizational forms and spatial linkages pertaining to recycling functions and external influential factors such as a feasible recycling technology and the existing industry structure. By conceptualizing organizational decisions, the theoretical model will suggest possible spatial patterns of recycling systems. In light of conceptualization, the research will investigate two different recycling systems of CDW recycling and waste carpet recycling. CDW recycling represents a localized recycling system with a traditional institutional approach of local government responsibility whereas waste carpet recycling is an exemplary case of an emerging institutional approach of manufacturer responsibility and an associated regional- or national-scale recycling system.

The second purpose of this dissertation is to construct a regional environmental IO model that quantifies the economic and environmental consequences of recycling industrial activities. The proposed IO model explicitly incorporates EOL management options by creating a separate industry and commodity sector for mixed CDW recycling, deconstruction, and carpet recycling. This feature enables the analysis of a closed-loop system of waste recycling in the IO framework. In addition, the research estimates the regional environmental coefficients in terms of energy use and greenhouse gas (GHG) emissions for assessing the environmental benefits of recycling systems. This regional environmental IO model will be utilized in two groups of simulations.

One simulation group demonstrates that the regional environmental IO model can be a useful tool that clarifies the environmental responsibility of a regional economy. Many environmental burdens that a regional economy manifests are associated with inter-regionally traded products and services, suggesting that the environmental emissions of a regional economy are driven by the demands of other regions; conversely, the consumption of imported products in a regional economy incurs environmental burdens of other regions. The environmental responsibility of a regional economy, thus, will vary according to the accounting principle that determines what types of emissions are attributed to what regions. This research will suggest a typology of the environmental responsibility of a regional economy pertaining to the origins of the supply and demand of products and then show single- and two-region environmental IO modeling frameworks that correspond to a typology of regional environmental responsibility. On the basis of the typology and regional environmental IO modeling framework, it will compute the extent of GHG emissions for which a regional economy is responsible in three metropolitan cases.

The second group of simulations of the regional environmental IO model analyzes the economic and environmental impact of four cases of recycling systems, including mixed CDW recycling, deconstruction, recycled nylon 6 production, and recycled carpet padding production. The research adds a new industry sector for each case of a recycling system in the framework of the two-region environmental IO model consisting of a metropolitan area and the rest of the nation. The simulations allow the evaluation of the economic and environmental consequences of specific recycling technology and associated collection systems in terms of output, income, employment, energy use, and GHG emissions.

Selection of Waste Materials and Research Areas

This research selects two types of waste materials: construction and demolition waste (CDW) and waste carpet. The recycling systems of two waste materials are significantly different in terms of institutional approach, the industry structure, and spatial patterns. CDW is often directly regulated by local ordinances. Environmentally proactive local governments have established diversion goals on construction and demolition projects, and local ordinances and local governmental support mostly drive a localized recycling system for CDW. Alternatively, waste carpet recycling is a case of voluntary agreements that involve the adoption of EPR policies. Self-regulation by responsible industry plays an important role in the development of a recycling system, and economic logic is a key determinant in the spatial linkage of recycling systems.

For building a regional environmental IO model, this research examines three metropolitan areas: the Atlanta metropolitan area, the San Francisco metropolitan area, and the Seattle metropolitan area. The metropolitan areas that consume and dispose of the majority of energy, material, and waste are the spatial foci of this analysis. The selection of case areas considered the local built environment, the economic structure, primary energy sources, and institutional contexts pertaining to carpet and CDW recycling.

The San Francisco metropolitan area is renowned for its proactive environmental policy of the diversion and recycling of CDW. Each local government within the metropolitan area has established a local ordinance that requires a certain level of CDW diversion on construction, renovation, and demolition projects. Hence, the rate of CDW disposal in landfills is considerably lower than that of other metropolitan areas, and the recycling infrastructure for CDW has been well developed in the San Francisco metropolitan area. It is germane to a case for theorizing the growth of a localized recycling industry and analyzing the economic and environmental impact of CDW recycling.

The Seattle metropolitan area has limited landfill space, and the tipping fee at landfills and transfer stations is one of the highest places in the U.S. Local and state governments have enacted and operated proactive environmental policies for recycling.

King County promoted a local policy that linked solid waste management and local economic development. For example, the Linkup Program aims at expanding the market for reusable and recyclable materials such as asphalt shingle, carpet, and gypsum board. In 2012, the State of Washington proposed a bill that creates a carpet recycling program based on producer responsibility.[1] These proactive institutional efforts lend themselves to the growth of recycling industrial activities worthy of investigation.

The Atlanta metropolitan area generated large amounts of CDW per capita because construction industry was a main driver of economic growth prior to the economic recession. Institutional support for recycling CDW has not been proactive in the Atlanta metropolitan area. Although construction activities have recently dropped because of the economic recession, substantial opportunities for CDW recycling still exist as the population continues to expand. In addition, within close proximity to Atlanta metropolitan area, northern Georgia is a national center of carpet manufacturing and also host to many carpet recycling-related facilities. Thus, the Atlanta metropolitan area is also an appropriate venue in which we can investigate the economic and environmental impact of carpet recycling.

---

[1] Washington State Bill 6341, accessed April 26, 2012, http://www.productstewardship.net/PDFs/carpet-factsheet-sb6341.pdf

Table 1: Comparison of Three Metropolitan Areas

	San Francisco Metropolitan Area	Seattle Metropolitan Area	Atlanta Metropolitan Area
Counties of Research Area	Alameda, Contra Costa, Marin, San Francisco, San Mateo	King, Snohomish, Pierce	Clayton, Cherokee, Coweta, Cobb, DeKalb, Douglas, Fulton, Fayette, Forsyth, Gwinnett, Henry, Paulding, Rockdale
Regional Planning Organization	Association of Bay Area Government: 9 Counties	Puget Sound Regional Council: 4 Counties	Atlanta Regional Commission: 10 Counties
Related Metropolitan Statistical Area	San Francisco-Oakland-Fremont, CA Metropolitan Statistical Area: 5 Counties	Seattle-Tacoma-Bellevue, WA Metropolitan Statistical Area: 3 Counties	Atlanta-Sandy Springs-Marietta, GA Metropolitan Statistical Area: 30 Counties
Demography			
Population 2009	4,317,853	3,407,848	4,749,461
Population Annual Growth Rate from 2000 to 2009 (%)	0.4	1.1	2.2
Land Area (Square Mile)	2,473	5,894	3,963
Construction and Residential Housing Permits			
Permit 2006	6,303	16,186	36,272
Permit 2009	2,392	5,396	3,157
Construction Cost 2006 ($ Millions)	3,007	4,557	6,652
Construction Cost 2009 ($ Millions)	939	1,425	690
Construction and Demolition Waste			
CDW Disposal	1.23 lb/day/person	1.80 lb/day/person	2.10 lb/day/person
CDW Landfill Tipping Fee	$69 -130/ton	$103/ton (Transfer Station)	$20.18/ton
Policy			
State Regulation	Local Government Construction and Demolition (C&D) Guide	Proposal of a Carpet Recycling Program (Producer Responsibility)	-
Local Ordinance & Support Program	Diversion Requirement (City of Alameda etc) Deposit Program (City of San Jose)	CDW Recycling Support Program (King County)	-
Link to Economic Development		Linkup Program (King County)	-

Source: Population: Population Estimates by County, U.S. Census Bureau; Land area: Land Area by County, U.S Census 2000; Construction permit and cost: Residential Building Permit, U.S Census Bureau; CDW generation: per capital CDW generation is calculated by total CDW generation divided by the population. Total CDW generation refers to waste. characterization study for each metropolitan area; CDW Policy: Government Agency Website, CDW diversion ordinance (http://stopwaste.org/home/index.asp?page=491), Linkup Program (http://your.kingcounty.gov/solidwaste/linkup/index.asp)

Contribution to the Field

This research can significantly facilitate the work of academic researchers and practitioners involved in local economic development planning. First, the theoretical model will provide a fundamental understanding of the location patterns of recycling facilities. It can facilitate the identification of local conditions that may contribute to the development of a local recycling industry. It will also inform local governments that seek to nurture the recycling industry about economic opportunities and environmental benefits that the local economy may reap. It may broaden the perspective on the recycling industry as a target industry for the local economic development. In addition, it is one of few studies that explore potential data sources of energy use and GHG emissions on the sub-national level. It will develop a regional environmental inventory connected to regional IO tables. Finally, the research presents an extended IO modeling framework for the recycling industry. The proposed model is flexibly applicable to other EOL management cases.

Overview of Research

The dissertation is organized as follows. Chapter 2 provides a comprehensive review of previous research on four topics, including theories of ecological modernization and industry organization, the geographical aspect of recycling, the technology of recycling, and the advance of the environmental IO model. Chapter 3 presents a theoretical model of the growth of the recycling industry and provides two cases of recycling systems in different institutional contexts. Chapter 4 describes the analytical tool of a regional environmental IO model extended to the recycling industry. Chapter 5 explores the data sources of energy use and GHG emissions and shows the procedures for estimating regional environmental coefficients by sectoral groups. Chapter 6 presents results that specify the amount of GHG emissions that each regional economy is responsible for in the case of three metropolitan areas. Chapter 7 describes simulated scenarios of the four cases of recycling systems and shows the results of simulations. Chapter 8 summarizes the policy implications for sustainable local economic development.

CHAPTER 2. LITERATURE REVIEW

The literature review aims at understanding four important dimensions pertaining to the development of theoretical and analytical models for the recycling industry. The key dimensions reviewed here are the evolving institutional context of waste management, the organizational and spatial patterns of the recycling industry, the technological aspect of recycling, and the advancement of a regional environmental IO model. The literature review chapter consists of four sections. The first examines a set of theoretical studies pertaining to environmental-led institutional and industrial changes and the spatial process and gives insights into which factors one should take into account when building a theoretical model of the growth of the recycling industry. The second section investigates the spatial dimension of recycling systems, highlighting the role of urban areas that support vital recycling systems. The third section provides fundamental information about the technological aspect of CDW and waste carpet recycling, and the final section reviews the history and recent advancement of environmental IO modeling.

2.1. Institutional and Industrial Change for Sustainable Local Economic Development

2.1.1. Evolution of the Concept of Local Economic Development

The concept of local economic development has evolved. The normative goal of local economic development has invited different perspectives on the status quo of social, economic, and environmental conditions. The historical changes in the concept of local economic development have been well documented in the literature. Fitzgerald and

Leigh (2002) initially addressed equity and environmental sustainability issues in the objectives of local economic development. In particular, they suggested that sustainable resource use and production should be considered of the principle of local economic development. Later, Patterson (2007) also conceptualized the evolving stages of local economic development approaches from traditional to progressive to sustainable local economic development. She considered sustainable local economic development as the next evolutionary phase. The traditional local economic development approach with agenda of business attraction and job growth conflicted with issues of equitable and environmental sustainability. In her paper, Patterson discussed a reconciled perspective instead of conflicting one between traditional and sustainable views.

Despite concerns about potential conflicts, the concept of sustainable development has already been embraced in some pioneering plans of local economic development. For example, the primary goal of the economic development plan of the City of Portland, Oregon, was to build a sustainable economy. Its industry cluster development plan targeted the clean technology industry such as solar and wind energy (City of Portland and Portland Development Commission, 2009). In 2009, the Chicago metropolitan region prepared a plan for green economic development strategies that contained several elements related to environmental sustainability in its local economic development plan. It intended to develop a metro-wide sustainability plan with strategies such as supporting renewable energy production, improving the energy efficiency of buildings, promoting waste reduction and resource conservation, and raising funds for green businesses (RCF Economic & Financial Consulting and Chicago Metropolitan Agency for Planning, 2009). The City of Toronto envisioned an economic development plan seeking to nurture green businesses that produce products and services that both directly and indirectly reduced environmental impact. It also discussed a strategy pertaining to how the city would compete with other cities in the clean and green industry sectors (Toronto Economic Development, Culture and Tourism, 2007). Overall, the terminology of green economic development, clean technology industry, sustainable economy, and resource and energy efficiency is commonly shared in their plans, which reflect a concept evolving toward sustainable local economic development.

2.1.2. Ecological Modernization

Although the theory of ecological modernization has not been widely discussed in the U.S., ecological modernization theory is an apropos starting point of the discussion of social change from recycling activities, the emergence of associated industry, and evolution of the regulatory approach (Schlosberg and Rinfret, 2008). Ecological modernization theory originated in western European countries, specifically, Germany and the Netherlands, in the mid-1980s, when the issues of environmental risk of industrialized societies were addressed and the direction of societal technological progress challenged (Cohen, 1997).

The theory postulates that the solution to the ecological crisis lies in the advance of technology and an industrial transition to ecologically rational organizations rather than the pursuit of the radical restructuring of deep ecology (Fisher and Freudenburg, 2001). While developed through different positions of authors and western European national contexts, the theory contains some core features: an optimistic perspective on the potential of technological innovation, a call for changes in environmental policy approaches, and the identification of a new role for non-governmental organizations (Mol, 1997 and 1999).

The theory of ecological modernization rejected the view of an adversarial, trade-off relationship between the environment and the economy and suggested an integrative approach that simultaneously achieves environmental and economic objectives within the existing capitalistic structure (Haughton and Counsell, 2004). It calls for an environmentally benign transformation of the industrial production system and organization, labeled an "ecological switchover" by Huber (cited in Gibbs, 2000). The motto of "pollution prevention pays" and a "win-win solution" are the founding premises of ecological modernization, which posits that innovation has a great market potential for meeting global environmental needs (Jänicke, 2008).

While early studies of ecological modernization were criticized for overemphasizing the role of technological innovation, later studies extended the theory by connecting industrial change to the policy-making process (Christoff, 1996). Hajer (1995) discussed why the notion of ecological modernization appealed to governments and why it had become a central issue of policy discourse in western European countries. Awareness of the failure of a command and control type of environmental policy was increasing in 1970s, which remedial measures and end-of-pipe control were the main approach. Ecological modernization theory was recognized as contributing to environmental discourse on the re-legitimization of the social regulation of environmental pollution. Since ecological modernization conceptualized environmental protection as a positive-sum game, regulation was not viewed as automatically conflicting with capitalistic logic and business interests. The characteristics of the regulatory style associated with modernization theory are communicative, democratic, open-ended, precautionary, and anticipatory (Christoff, 1996).

The theory also suggested the promise of new markets and a demand for environmental goods, so the alternative conceptual language of ecological modernization was widely diffused in the environmental policy-making process in western European countries. Mol (1999) contended that diverse political environmental programs could be interpreted within the ecological modernization framework, arguing that the increasing use of economic instruments in environmental policy and a consensus-based environmental policy can be interpreted as the adoption and the diffusion of ecological modernization ideas in the policy-making process. Jänicke (2008) found that “soft” regulation, such as voluntary agreement, has often been insufficient. He envisioned an innovation-friendly framework of environmental regulation that imposes a strict standard that eliminates uncertainty and risk and at the same time provides economic incentive that drives innovation.

Finally, ecological modernization theory emphasized the role of non-governmental organizations in the ecological transformation of society. For example, Sonnenfeld (2002) illustrated that non-governmental organization played an important role in developing cleaner technology in the pulp and paper manufacturing. His example illustrated that some environmental movements cooperating with both government and industry may promote more reform-oriented change that does not require radical opposition to capitalistic logics

2.1.3. Industry Organization

Although industrial organization was disregarded in classical location theory mainly built upon transportation costs, it is a key determinant in the location patterns of industry (Scott, 1983). A subsequent theory of the firm focused on organizational change in the vertical integration and disintegration of firms and associated spatial implications. Industrial organization is viewed as a system of intra- and inter-firm transactions and a matter of economizing transaction costs (Scott, 1983, Williamson, 1985). This theory drew from the seminal work of Coase (1937), which sought to explain the balance of boundary between internally coordinated firms and the externally transacted market: that is, whether products are manufactured within a very large firm or in numerous linked firms. The balance is determined by two types of transaction costs: internal coordinating costs and external transaction costs. Under simplified conditions, when the bureaucratic coordinating cost in the integrated production system exceeds the cost of purchasing products from the outside, a vertically-disintegrated production system will occur. Conversely, when external transaction costs exceed coordinating costs, a vertically- or horizontally-integrated firm will emerge.

In an effort to examine the organizational form of industry, Williamson focused on asset specificity as a key factor that determines whether the cost of internal production or external transaction is greater. Asset specificity refers to “the degree to which an asset can redeployed to alternative uses and by alternative users without sacrifice of productive value” (Williamson, 1989, pp. 142). Williamson argued that when investment is more specific to a particular buyer-seller relationship, internal production or long-term contracts are more likely to occur because they do not incur costs associated with opportunism and uncertainty occurring in the bargaining process or from market contracts (Joskow, 1988). Asset specificity can take different forms: site specificity, meaning that once an asset is invested, it is highly immobile; physical asset specificity, meaning that equipment is designed for a specific transaction and that another use may lower its value; and human asset specificity, which arises from a learning-by-doing process (Williamson, 1989; Joskow, 1988).

From the perspective of industry organization theory, the spatial pattern of industry and industrial organization forms are co-dependent (Wood and Parr, 2003). Scott (1983) predicted that a disintegrating group of firms has a tendency to spatially converge in order to economize on spatially-determined costs. The unit cost per transactional activity will be lower in a dense large scale of a network. Alternatively, in the multi-establishment firm with integrated functions, branch plants are more easily spatially dispersed because of stable supply relationships and well-established functional roles coordinated by higher order of office. The location of internal production units is loosely confined inside of a wider region.

2.1.4. Implications

The preceding literature review highlighted the environmental-led transformation of industry production tied to a new approach of environmental policy as a real, on-going phenomenon. Studies suggested that change toward sustainable production and the emergence of closed-loop systems can be understood as a reciprocal process of building up a new societal institutional rule and new production practices among regulatory agencies and industry. In addition, they have suggested that organizational change resulting from a reaction to new sustainable production and closed-loop systems may be associated with new spatial patterns. Thus, this implies that the growth of the recycling industry should be investigated through the lens of an integrative framework that accounts for institutional changes, industrial and organizational decisions, and spatial processes. The previous literature has seldom articulated how recycling industry grows in various institutional settings or how it is spatially organized in the regional economy. Under a condition of under-theorization, this research seeks to develop a theoretical model that explains the spatial growth pattern of the recycling industry in different institutional settings.

2.2. Geography of Waste Recycling

2.2.1. Industrial Symbiosis and Planning

The concept of industrial symbiosis stemming from industrial ecology has drawn considerable interest and been introduced to the planning field (Dunn and Steinemann, 1998; Clinton, 1999). Industrial symbiosis is defined as "wastes from one industrial process can serve as the raw materials for another, thereby reducing the impacts of industry on the environment" (Clinton, 1999, p. 365). Because industrial symbiosis is a place-based phenomenon, it has received attention in the planning literature (Derochers, 2001). For example, in a small industrial town, the city of Kalundborg, Denmark, several bilateral agreements were made among various entities—a power plant, an oil refinery, a wallboard manufacturer, a pharmaceutical company, a fishery, and the city government. One of the key binding elements among these seemingly unrelated entities was geographical proximity. This case inspired attempts at discovering and establishing other cases of symbiotic relationships and to integrate them into the planning process.

The suggested planning tool was the eco-industrial park development in which planned symbiotic waste exchange was expected to occur. It was promoted by the U.S. President's Council of Sustainable Development (PCSD) (1999). According to the PCSD, the eco-industrial park was "an industrial system of planned materials and energy exchanges that seek to minimize energy and raw material use, minimize waste, and build sustainable economic, ecological and social relationships."[2] Dunn and Steinemann (1998) emphasized a potential role of planners in the development of eco-industrial parks, suggesting that planners can play a role in coordinating communication among industry, the government, and a local community for the purpose of initiating waste material exchange. Deutz and Gibbs (2004) envisioned a planned eco-industrial park as a means of place promotion. They hypothesized that the economic benefits generated from waste material exchange and the cleaner environment may help to attract new business and capital investment to a planned eco-industrial park.

Previous studies surveyed and evaluated actual cases of eco-industrial park development. Heeres et al. (2004) examined three eco-industrial parks in the Netherlands and three in the U.S. They found that eco-industrial park development in the U.S. was more likely to be driven by local governments than that in the Netherlands. Despite the support of local governments, the U.S. cases were less successful because companies were less actively involved in waste exchange. The authors argued that the core element of the successful development of a symbiotic relationship lies in a proactive role of a company leading the initiative in the development of an eco-industrial park. Gibbs et al. (2005) performed a comprehensive survey on eco-industrial

[2] Website of *Eco-Industrial Park Workshop Proceedings,* accessed on November 4, 2010 at http://clinton2.nara.gov/PCSD/Publications/Eco_Workshop.html#iv

development in Europe and the U.S. They found a disjunction between the theory and the practice of eco-industrial development. They pointed out that as an economic development tool, the actual plan of eco-industrial development consisted of multiple components such as a symbiotic relationship, a recycling business cluster, life-cycle assessment, job training, deconstruction and remanufacturing, green product design, environmental innovation, and public participation and collaboration. They identified 35 cases in the U.S., only six of which were operational, nine planned, and sixteen classified as attempted. They argued that the evidence of symbiotic waste exchange was rare in the U.S., and waste exchange could not be a sole viable strategy of eco-industrial development. Indeed, the extent of waste exchange was only one measure of the success of eco-industrial park development. Chertow (2007) investigated a planned and self-organized industrial symbiotic relationship. She found that most mutually beneficial waste exchange was self-organized rather than designed, planned, and developed and concluded that the existing economic advantage, the industrial structure, and the supply chain should be considered in policy development for waste exchange and that planning and policy should focus on identifying precursors and enhancing the existing symbiotic relationship.

2.2.2. Role of a City in Waste Recycling

The issue of the spatial scope of waste exchange was criticized with regard to eco-industrial parks with a narrow geographical scope. The literature has investigated the potential of cities and regional industrial districts in which diverse and plentiful industrial activities are rooted. Desrochers (2001) reviewed the historical archives that documented industrial recycling linkages and found that considerable by-product recovery and recycling occurred within cities through intra- and inter-regional channels. His findings implied that cities historically played an important role in building inter-industry recycling linkages. He also argued that eco-industrial development was not necessarily restricted to narrow spatial boundaries and that inter-industry recycling linkages would emerge on the regional level. Lyons (2007), who empirically studied the geographical scale of recycling, remanufacturing, recycling manufacturing, and waste treatment firms in Texas, argued that economic logic dominates the geographical scale of recycling and that different industries have different geographical scales of material and waste transactions. Classifying recycling-related industries into local cycling, exporting cycling, and multiple geographical scale cycling, he conducted a survey that found that local recycling plants are likely to be located near areas where input sources are generated because input material with relatively low economic value is too costly to ship. He concluded that the spatial dimension of recycling linkages can be local, regional, national, and global, implying that economic logic should be considered a dominant factor in determining the spatial dimension of recycling linkage. Sterr and Ott (2004) argued that the regional scale is more suitable for developing a symbiotic relationship among firms; however, they pointed out that coordination and trust, important factors that promote symbiotic relationships among industrial entities, may be challenged on the regional level. The authors contended that diverse interests, low levels of organization, and lack of trust among firms necessitate a suitable planning instrument on a regional level.

2.2.3. Environmental Responsibility of a City

Numerous urban production and consumption activities exert substantial environmental pressure. A United Nations report mentioned that urban areas are responsible for 75% of global energy use and 80% of GHG emissions (cited in Dodman, 2009). The role of cities and urban areas in climate change and GHG emissions has received attention (Hoornweg et al., 2011). A number of studies focused on quantifying GHG emissions on a sub-national scale. For example, the International Coalition for Local Environmental Initiative (ICLEI-Local Government for Sustainability) developed an accounting method of GHG emissions on the sub-national level, called the International Local Government Greenhouse Gas Emissions Analysis Protocol (IEAP). The IEAP recommended that the city-level GHG inventory include three scopes of emission sources: Scope 1 consists of direct emissions from fuel combustion within the boundary of a city; Scope 2 contains indirect emissions from purchased energy such as electricity and district heating generated out of a city but consumed within a city boundary; Scope 3 includes all other indirect and embodied emissions originating from products consumed within a city boundary but produced out of a city. In actual studies of the city-level GHG inventory, neither the accounting method nor the scope has been standardized. Kennedy et al. (2009) surveyed the method and scope of the sub-national-level GHG inventory studies. They investigated the sub-categories of emission sources included in GHG inventory studies: direct emissions from energy, waste, industrial process, agriculture, forestry, and land-use, and cross-boundary indirect emissions. They found that few studies included Scope 3, cross-boundary indirect emissions.

Although cross-boundary emissions are omitted in many studies, they may be significant sources of GHG emissions for a city in which many industrial activities rely on imported energy and products. Hoornweg et al. (2011) showed that urban GHG emissions per capita are generally lower than the national average in most developing and developed countries except China. In the U.S., the national average GHG emissions was 23.59 tons of CO_2 Eq. per capita, but major cities' GHG emission such as Boston, Chicago, Dallas, Los Angeles, New York City, Seattle, and San Francisco ranged from 10 to 15 tons of CO_2 Eq. per capita.

The disparity between national and urban GHG emissions could be the result of two factors. First, urban environments allow more energy-efficient lifestyles rather than non-urbanized environments (Brown et al., 2008). The second reason is associated with the economic structure of the urban economy and the practical accounting method and scope. Since many U.S. cities have already shifted to service-based and consumption-oriented economies, if the GHG emissions inventory study had omitted cross-boundary GHG emissions, per capita GHG emissions of cities may have been significantly lower than those nationally. Dodman (2009) argued that the omission of cross-boundary GHG emissions can considerably distort the environmental responsibility of a city. The amount of energy requirements and GHG emissions for fulfilling a function of a city may be greater if energy use and GHG emissions associated with traded products crossing the boundary are taken into account. Hillman and Ramaswami (2010) estimated the amount of cross-boundary emissions contributing to per capita GHG emissions for eight cities in the U.S. They computed GHG emissions from inter-regional air and freight transportation and embodied energy of selective products such as water, food, and cement. Finding that cross-boundary GHG emissions comprised an average of 47% of emissions in eight cities, they concluded that the omission of cross-boundary GHG emissions resulted in significant bias.

2.2.4. Implications

This review shows the importance of geographic scope in the development of recycling systems. Waste exchange and recycling are not necessarily confined to small, planned industrial parks. Recycling systems are more likely to be successfully built on urban and regional economies. In addition, economic logic is a key factor that determines the geographic scale of market-based recycling systems. The literature implies that an urban or regional economy would be an appropriate spatial unit of analysis, but a geographic scope of recycling linkage extends beyond the regional economy when recycled materials are valuable enough to ship a long distance. Therefore, while the metropolitan area is a basic spatial unit of analysis, research sometimes investigates recycling systems from a wider spatial perspective, particularly in the case of carpet recycling.

The literature review, which also revealed that cross-boundary GHG emissions in the sub-national GHG inventory occupy a significant portion of emissions, suggested that a sub-national GHG emission study requires an accounting method capable of computing embodied energy of imported and exported products and associated GHG emissions. The regional environmental IO model can be utilized as a useful tool that measures the cross-boundary GHG emissions. Thus, this research will construct single-and two-region environmental IO models, which will enhance our understanding of the environmental responsibility of a city.

2.3. Material Flow and the Technology of Recycling

2.3.1. Urban Sustainability and Construction and Demolition Waste

The composition and quantity of waste materials are key pieces of information for building a plan of recycling-based sustainable local economic development. That few statistics measuring the stock and flow of EOL products have been available necessitates an analytical method that computes the volume of a certain waste stream at a sub-national scale. Leigh et al. (2007) proposed a method of estimating the stock and flow of waste on a metropolitan scale involving a case of obsolete computers by examining spatial and demographic differences among such variables as computer ownership and computer lifetime in households and industries.

The stock and flow of building materials has also been a core research topic of in the evaluation of urban sustainability. Haberl et al. (2004) argued that the stock of buildings and infrastructures is relevant to urban sustainability because the conversion to urban land use directly affects the productivity of the biosphere, and maintaining and re-using material stocks affect the quantity of virgin materials required to meet current demand. In other words, how a society uses and maintains its stock of buildings and infrastructures is directly associated with the use and the flow of future resources. An urban metabolism approach that reveals how much energy, waste, and material circulate in and out of cities provided an analytical framework of urban sustainability (Sahely et al., 2003). In light of the perspective of urban metabolism and the importance of urban building and infrastructure stocks, Huang and Hsu (2003) analyzed the embodied energy of construction materials in Taipei, and they showed that the embodied energy of construction material consisted of 46% of total embodied energy. Gao et al. (2001) examined the amount of energy saved by using recycled materials instead of virgin materials in construction and found that recycled material use contributed to energy savings of at least 10% comparing to use of virgin materials in a case of Japan.

The U.S. EPA (2003) estimated the quantity of building-related CDW generated in the U.S. The sources of CDW generation fall into six groups based on building type and life cycle: residential construction, non-residential construction, residential demolition, non-residential demolition, residential renovation, and non-residential renovation. The equation employed to estimate the amount of CDW is relatively straightforward: The total amount of CDW is calculated through multiplying the total industrial activities of new construction, renovation, and demolition by some CDW generation factor such as tons of CDW per million dollars or per square foot. According to U.S. EPA estimates, CDW generated from demolition activities accounted for about 50%, that from construction about 9%, and that from renovation about 41% of the total. The weight of CDW per unit varies across regions where material use patterns, building styles, building stock age, and construction activities significantly differ. Cochran et al. (2007) estimated building-related CDW generation in Florida in the same categories used in the U.S. EPA study. The results showed that the amount of CDW per capita in Florida was less than that of the national average. The authors found that high construction and low demolition activities contributed to the lower CDW generation rate in Florida.

The waste composition of the CDW stream was mostly surveyed by the environmental departments of state governments or counties through direct sampling in a field survey and statistical estimation (California Integrated Waste Management Board, 2006; Georgia Department of Natural Resources, 2010; King County Department of Natural Resources and Parks, Washington, 2009). Although the composition of waste materials in the CDW stream cannot be consistently compared because of diverse classification and sampling methods, the major waste materials of the CDW stream are concrete, asphalt roofing, wood, and drywall.

2.3.2. Deconstruction

The demolition of obsolete building stocks accounted for half of U.S. CDW generation (U.S. EPA, 2003). An alternative building demolition technique that allows the maximization of the potential of CDW recycling is deconstruction, defined as "the process of carefully dismantling a building in order to salvage components for reuse and recycling" (Leroux and Seldman, 2000, p. 1). While traditional demolition is a mechanized, capital intensive process, deconstruction is a more labor intensive process. This feature of the deconstruction process potentially provides job opportunities for the low-skill labor force in a community. Hence, it has also been recognized as a means of local economic development (Leigh and Patterson, 2006).

Deconstruction can be classified into non-structural and structural (Housing and Urban Development, 2001). Non-structural deconstruction, which typically occurs during a renovation process, involves the removal of non-structural components such as floor finishes, cabinetry, windows, and fixtures for salvaging building components.

Structural deconstruction is the dismantling of building components, including concrete frames and wood, lumber, brick, and roof systems, which contribute to structural integrity. Deconstruction is considered to offer valuable economic benefits because it is responsible for salvaging valuable building components, reducing heavy equipment costs, and creating jobs for local communities. However, the use of this approach has been limited by a number of factors such as the timeline constraints of new construction projects, policies pertaining to housing preservation, and immature markets for salvaged building materials (Leroux and Seldman, 2000), so it is not in widespread use. In fact, the U.S. Department of Housing and Urban Development (2001) assessed that structural deconstruction was feasible only in metropolitan areas that contain a large number of abandoned and obsolete buildings.

2.3.3. Construction and Demolition Waste Recycling

Although major construction materials are globally plentiful, the supply of some may be insufficient at local and regional levels (Horvath, 2004), which may require the transportation of heavy construction materials a long distance inter-regionally. Such transport may result in increased costs of construction material. To meet local demand for construction materials, an alternative to costly transport can be recycled CDW materials. Unfortunately, the growth of the CDW recycling market has been hampered by inferior physical properties of recycled materials, government regulations, inconsistent market supply and demand, lack of a collection and processing infrastructure, and fluctuation of recycled material prices. This section reviews the potential for and barriers to recycling the major CDW materials: concrete, drywall, wood, and asphalt shingles.

Of the four major materials, concrete waste is the heaviest material in the CDW stream, so the recycling of these materials could substantially reduce the costs of transportation. CDW processors typically have mobile recycling equipment for on-site recycling, which can reduce transportation trips and associated costs (Robinson et al., 2004).In light of the high cost of tipping fees at CDW landfills and transportation, concrete recycling is economically more practical. Concrete waste can be transformed into recycled aggregate through the simple processes of crushing and separating it from metal and other debris and screening fractions by size. Recycled aggregate is used as a sub-base in road construction, general fill, asphalt concrete, and cement concrete. In fact, the most common end-use of recycled aggregate is as a sub-base in the road construction while a relatively small portion is used in asphalt and cement concrete (Kelly, 1998).

Although pursuing an end use with higher economic value is desirable, some technical and institutional barriers do not allow the use of recycled aggregate in cement concrete production. Technically, the material property of recycled aggregate does not have as much integrity as that of natural aggregate. In recycled aggregate, cement debris clings to gravel, which lowers the density of the aggregate; and cement debris absorbs more moisture than natural aggregate, leading to more shrinkage in cold weather, more expansion in hot weather, and ultimately more cracking (Robinson et al., 2004). The compressive strength of concrete with recycled aggregate is typically less than that of concrete with natural aggregate. For this technical reason, only qualified recycled aggregate is used for structural purposes (Wilburn and Goonan, 1998). However, a recent study showed that a mixture of recycled and natural aggregates can produce cement concrete with the compressive strength similar to that of natural aggregate concrete (Marinkovic et al., 2010).

Since drywall is popularly used for interior purposes, it makes up a large share of the CDW stream. Drywall is made of gypsum, which causes an odor problem when it is decomposed in landfills. Waste drywall is generated during both new construction and demolition projects, the latter generating about 90% (King County Department of Resources and Parks, 2006). The process of drywall recycling is simple. The paper that covers gypsum is removed, and the gypsum is ground up for use as feed stock. According to a study of drywall recycling (Orange County and Seminole County, Florida, 2003), waste drywall has several potential end uses in agriculture (i.e., the mitigation of subsoil acidity, improvement of the soil structure, nutrients for plants) and construction (i.e., new gypsum and aggregate in concrete). Although drywall recycling causes no technical problems, recycled drywall is not competitive in the market (Massachusetts Department. of Environmental Protection, 2008). Reasons for this may be that the price of virgin gypsum material is very low, and synthetic gypsum generated from power plants as a by-product is supplied to the market at half the price of virgin gypsum.

CDW typically contains several types of wood waste such as dimensional lumber, pallet, crate, engineered wood, painted wood, roofing and siding, and furniture. Wood waste in the CDW stream has large potential for recycling in various ways (Tam and Tam, 2006). First, dimensional lumber from old buildings is a valuable material in re-use for structural purposes. It is simply re-processed through cleaning, de-nailing, and sizing. However, it should be separated from mingled CDW or contamination. In addition, chipped and grinded waste wood has several applications. A common outlet of chipped wood waste is bio-fuel, which has a large tolerance for contaminants such as painted and stained wood (King County Department of Resources and Parks, 2006). The demand for waste wood for bio-fuel was assessed to be plentiful in the northeastern states (Massachusetts Department. of Environmental Protection, 2008). Chipped wood waste is also frequently used for landscape and compost. Another potential use of wood waste is the production of pulp chip, an alternative to virgin pulp. Several processors in Washington sell recycled-wood pulp chip. However, this process requires a high grade of wood, and engineering wood waste is not acceptable because recyclers have to know the original species of wood (King County Department of Resources and Parks, 2006).

Asphalt shingles, a major roofing material comprised of a large quantity of CDW material, have potential use in hot mix asphalt, in which recycled asphalt shingles are blended with asphalt pavement and then used for paving roads. In addition, they can be utilized for pavement cold patch used for filling potholes and small-scale sidewalks and utility cuts, and aggregate road base. However, the current market demand for recycled asphalt shingles is not active (King County Department of Resources and Parks, 2006).

2.3.4. Carpet Recycling

The amount of discarded carpet comprises roughly 1% of municipal solid waste in the U.S. (Guidry, 2008).Although the volume of waste carpet is smaller than other CDW materials, waste carpet has substantial potential for recycling in terms of physical property and economic value. Annually, five to six billion pounds of waste carpet have been discarded from residential and commercial property in the U.S., and the diversion rate was approximately 5.6% (Carpet America Recovery Effort [CARE], 2010). Hence, abundant reusable and recyclable materials such as nylon 6, nylon 6.6, and polypropylene currently end up in landfills.

Four options are available for waste carpet: reuse, waste-to-energy, primary recycling, and secondary recycling (Guidry, 2008). The lifespan of carpet can be extended through the reuse process of collecting, cleaning, and reconditioning. However, the portion of re-use is relatively small. Waste carpet can be used as fuel in cement kilns and waste-to-energy facilities. According to the 2010 CARE survey, more than ten percent of diverted waste carpet is burned as a fuel. The waste-to-energy option can reduce fossil fuel consumption, but the generation of harmful end products such as ash and noxious gas is a challenging issue (Wang et al., 2003).

Waste carpet can be recycled in two ways. In primary recycling, materials in waste carpet are recovered in their original form and used for a similar physical function. For example, the nylon 6 in the face fiber of waste carpet can be transformed back into recycled caprolactam, a feedstock used for producing recycled nylon, and the recycled nylon 6 can be an input of a recycled-content carpet product. Hence, the primary recycling option can be a pure closed-loop recycling system in which materials in waste carpet are repeatedly recovered and used in new carpet products while maintaining their original physical function. Another common form of waste carpet recycling, the secondary recycling option, is open-loop recycling, in which materials in waste carpet are reclaimed and used in different forms of products. Waste carpet materials are converted into plastic pellets that are potentially used in various types of plastic products

(Lave et al., 1998). Several types of products that use reclaimed engineering resins from waste carpet are sold in the market (CARE, 2010).

2.4. Input-Output Model Related

2.4.1. Environmental Input-Output Model

The IO model, a major tool of traditional local economic development planning, has been widely used in economic impact analyses applied to various cases such as manufacturing facility relocation, direct foreign investment, sports event hosting, and infrastructure investment. In addition to this primary role, researchers have attempted to extend the IO framework in order to encompass environmental issues, and have demonstrated that the IO model is a flexible platform on which environmental problems and issues can be effectively analyzed. Previous studies pertaining to environmental IO modeling are summarized in three aspects: 1) The IO model was utilized to account for the material flow and energy use in the economy. It investigates how much energy and materials go in and out of a local economy and what types of materials circulate among economic entities; 2) the IO model modified, being capable of analyzing pollution and waste generation and treatment, was applied to environmental planning and solid waste management; and 3) the IO model was used as a platform for inter-disciplinary research. It has been adopted in the field of industrial ecology, taking root as an analytical tool in life-cycle assessment (LCA).

Previous research has shown that the monetary unit and physical unit IO models are compatible. Because economic data are typically compiled according to monetary values, the benchmark IO tables published by the U.S. Bureau of Economic Analysis every five years are always constructed on monetary transactions. Duchin (2004), however, argued that the monetary unit IO model is a special case for measuring economic transactions. She provided a numeric example of the duality between the physical quantity IO model and the price IO model. Whereas actual IO modeling cases measured by physical units are not common, some European countries such as the Netherlands, Germany, Denmark, and Finland have attempted to develop physical IO tables (Hoekstra and Bergh, 2006).

The enterprise IO model is an example of a physical quantity IO model. Lin and Polenske (1998) developed an IO table for a single plant that recorded the flow of materials with physical units among processes and from the market. The model displayed the flow of input material consumed in the process, and the output, the by-products, and the waste produced in the case of iron and steel production. Albino et al. (2003) extended the enterprise IO of a single company to the industrial district, an agglomeration of small-size firms. They surveyed how local companies in an Italian industrial district specializing in the production of leather sofas were networked through energy and material flows. The model traced material transactions from one company to another, energy input into the process, and the output and by-products of the processes. Pedersen and Haan (2006) provided a generalized form of physical flow accounting of input-output tables. These previous studies showed that the physical quantity IO model can be developed in an actual case.

The second extension of the IO model incorporated issues of resource management, pollution control, and solid waste management. An early issue that was integrated into the IO model pertained to pollution control (Leontief, 1970). Leontief's seminal work illustrated how the IO model could be extended through creating a new account for pollution and the industry of pollution abatement. In his model, the inter-industry table is modified by adding both a new row for pollutants generated from each sector and a new column for a pollution abatement sector. In subsequent studies, this modeling approach continued to be used in the issue of the management of municipal solid waste, including household and industrial waste. Huang et al. (1994) developed an IO model applicable to solid waste management. The model introduced R3 industries—recycling, reuse, and reduction—to the IO framework and also established ecological input and output accounts. The authors analyzed how much ecological output and input was emitted and consumed in R3 industries and showed that the IO model with R3 industries could be used for simulations that determined a preferable strategy of solid waste management.

Along with Leontief's initial framework and an extension of Duchin's work, Nakamura and Kondo (2002) proposed a waste input-output (WIO) model in a case of the Japanese economy. The WIO model analyzed the impact of various cases of waste management such as the recycling of electrical appliances, the life cycle costs of appliances with different energy efficiency, and the life cycle management of polyvinyl chloride (Nakamura and Kondo, 2006a; Nakamura and Kondo, 2006b; Nakamura et al., 2009). Pimenteria et al. (2005) developed a sub-national IO model that incorporated the circular flow of recycled materials within a local economy. To analyze the economic and environmental impact of recycled materials that are re-introduced into an economy, they added recycling and collection industries to the inter-industry table. They also conducted a simulation that examined energy savings and the reduction of CO_2 emissions in the state of Rio de Janeiro.

Within the IO framework, Steenge (1999) discussed the institutional aspects of a pollution problem pertaining to the issue of the party responsible for pollution. In an IO table, he illustrated how pollution and its related costs could be imputed to the industry sector (i.e., the polluter) or the household (i.e., the user), depending on the guiding principle of whether the polluter or the user pays. For resource management and future growth scenarios, Lange (1998) developed an IO model accounting for natural resource consumption and pollutant emissions and used it to predict the course of future development of the Indonesian economy. The simulation model examined whether the Indonesian economy would grow without resource constraints and serious environmental degradation.

Finally, along with its development in the economics and planning fields, the IO model has recently been advanced in the field of industrial ecology (Suh and Kagawa, 2005), a field intended to promote closed-loop industrial systems through building symbiotic relationships (Lifset, 2009). The disciplinary tradition emphasizing the systems view and the commonality between industrial ecology and the IO approach has led to the adoption of the IO model in industrial ecology research. In particular, LCA, which aims to analyze a full range of environmental effects of a product or a process, utilized the IO model as an analytical tool. The next section reviews the advancement of IO modeling in industrial ecology in detail.

2.4.2. Input-Output Model in Life Cycle Assessment

The IO model has been recently employed as an alternative analytical tool in a LCA study. LCA is a methodology that assesses the lifetime environmental impact of a product from production, use, and disposal (Rebitzer et al., 2004). It quantifies all material flows, i.e., the input resources, by-products, and emissions of a product within associated production systems. LCA is often used for comparing the environmental impact of the two alternative material uses with improvements in the environmental performance of a product.

The LCA study typically started from a microstructure of an economic system, which is a combination of various processes associated with a studied product (Suh, 2004).The first step of LCA is to define the unit of analysis, called a "functional unit," and to draw a product system boundary for implementation of LCA. The system must include processes that significantly contribute to a studied product and that are expected to be affected by the studied product (Rebitzer et al., 2004). After that, all relevant data about resource input and environmental output are compiled. This step is called the "inventory analysis." Then, it conducts an evaluation of the environmental impact of a product. This LCA approach is called a "process-based LCA model."

The IO model was recognized as an alternative LCA tool to a process-based LCA model (Suh and Kagawa, 2005; Suh and Huppes, 2009), for the selection of a system boundary is a serious practical problem for LCA practitioners in process-based LCA analysis (Suh et al., 2003). Inventory analysis is a difficult and time-consuming task if an LCA practitioner encompasses all indirect upstream relationships. Hence, a LCA practitioner has to decide the boundary of systems investigated in a LCA study by cutting off the upstream and downstream relationships in certain level. The cut-off, however, may be subject to subjective judgment and generate a systematic truncation error. Lenzen (2000) argued that when a process-based LCA approach considers only first-order inputs, the rate of truncation errors for most commodities rose to more than 50%. Regarding the system boundary problem, the IO model clearly has an advantage in that it already contains all industry sectors of the entire economy, and it allows all flow of goods and services to be traced among industry sectors. Therefore, for the purpose of avoiding truncation errors, the IO model has become complementary to the process-based LCA model (Lifset, 2009; Suh and Huppes, 2009; Suh, 2004).

The Green Design Institute developed the economic input-output-life cycle assessment (EIO-LCA) model for the U.S. economy for applied analysis. Hendrickson et al. (1997) showed that the result of an LCA study using the EIO-LCA model was similar to that using the process-based LCA model. The EIO-LCA model was built upon plentiful energy and environmental datasets collected in the U.S. This model includes criteria pollutant emissions (SO_2, NO_x, particulate matter, CO, volatile organic compounds), toxic pollutant emissions that are retrieved from the Toxic Release Inventory (TRI) database, GHG emissions (CO_2 and ozone-depleting chemicals), and electricity and fuel consumption. The EIO-LCA has been applied to various cases such as the demand-based quantification of local GHG emissions on the metropolitan scale (Ramaswami et al., 2008), energy use and GHG emissions in the service industry (Rosenblum et al., 2000), the environmental impact of alternative material use in the case of automobile fuel tank systems (Joshi, 1999), and the environmental impact of construction material in the road and energy use of residential buildings (Hendrickson, 2006).

Previous studies (Suh and Huppes, 2009; Joshi, 1999) summarized the methods that connect the IO model to LCA: approximation, addition or disaggregation, and integration. The first, approximation, is a simple method. A single industry sector classified in the IO model may consist of sub-industrial groups that produce similar but heterogeneous products. It can lead to a “product-mix problem” (Miller and Blair, 2009). The industry classification of the IO model is often not matched to a product or a process that a LCA study defines and examines. The approximation method simply assumes that an industry sector classified in the IO model approximately represents the real production structure of a product that an LCA study investigates even though the IO industry sector might produce a different mix of products. While approximation is an easy and quick approach, the obvious disadvantage is that when an approximated industry sector may not well reflect the production structure of a studied product, it may result in large estimation errors. In addition, this approach cannot apply to atypical or new products.

The second method, addition or disaggregation, can avert a product-mix issue associated with the approximation approach. In this method, a new industry sector of a studied product is added or disaggregated from an existing industry classification of the IO model in order to adjust a different level of specification. Creating a new industry sector, precisely showing the input and output structures of an investigated product, requires additional information. For example, the U.S. benchmark IO model is not ideal for an LCA study of an EOL product because various waste management industrial activities are aggregated into a single sector (Choi et al., 2011). The addition or disaggregation approach can be useful in the analysis of EOL products because it may specify several waste management options such as collection, recycling, remanufacturing, incineration, and disposal in the IO tables.

The final approach is integration, which systematically combines process- and IO-based LCA models. Suh (2004) proposed an integrated framework. He conceptually adjusted an LCA framework with a functional flow-by-process in the context of a traditional IO framework and then plugged the information of process-based LCA into the larger framework of the IO model.

Recent developments of the IO model in industrial ecology are helpful to planning of sustainable local economic development. The advance of these approaches has extended the frontier of the IO model. The process-based LCA approach typically examines current and future production technology and associated energy, pollutant, and material use. It adds flexibility to the highly aggregated IO model by providing information of specified industrial processes. In particular, the existing IO industry classification is not applicable to recycling-related industries and secondary material flow (Jackson et al., 2008). The recently advanced approach can help us investigate the economic and environmental consequences of waste management on the local economy when detailed information of waste management is explicitly incorporated into the IO model.

2.4.3. Regional Environmental Input-Output Model

Substantial research effort has been dedicated to the development of the regional IO tables (Jackson, 1998). Whereas research on regionalizing inter-industry tables was abundant, few studies have examined the regionalization of environmental and energy use accounts owing to the dearth of relevant statistics at a sub-national level. In the United Kingdom, several projects initiated for constructing regional environmental accounts connected to a regional inter-industry table, such as the environmental input-output (ENVIO) table constructed for the Welsh economy (Munday and Roberts, 2006). Turner (2006) discussed a method of generating regional-specific fuel-use and pollutant emission coefficients. He compared a bottom-up approach based on locally surveyed data and a top-down approach that adjusted for national coefficients. In a case region of Jersey in the United Kingdom, results showed large variations in fuel use between region-specific and nationally-adjusted models. The paper argued that to create valid regionalized fuel use and emission coefficients, one must identify the various technologies that determine the extent of fuel use and pollutant emissions from region to region.

Cicas et al. (2007) proposed a regionalized EIO-LCA model at a state level in the U.S. The model regionalized inter-industry coefficients using the gross state product (GSP) and narrowed down the number of industry sectors from 491 in the national EIO-LCA model to 63 because of the more aggregated industrial classification of GSP data. Methods of regionalizing environmental factors slightly differ among the types of emissions according to data availability. If region-industry-specific statistics are available, emission factors are directly calculated from available data. For example, the state-level toxic releases are directly retrieved from the U.S. EPA's Toxic Release Inventory (TRI). When only aggregated region-specific data are available, national-level industry-specific data are used to allocate aggregated region-specific data. The electricity is a case that total state electricity consumption is disaggregated into industry sectors.

Finally, Lenzen and his colleagues constructed a multi-region input-output (MRIO) model for energy and environmental analysis for the Australian economy (Lenzen et al., 2004; Munksgaard et al., 2005; Lenzen and Peters, 2010). The model examined how demand in one region affects the environment of another region in terms of GHG emissions and water use. This model is potentially applicable to multi-regional planning.

2.4.4. Implications

This review shows that the development of the environmental IO model involved incorporating the issues of pollution, resources, and waste management. It has been flexibly adopted in both the economic and environmental planning fields. The recent nexus to the industrial ecology field has led to the development of the national environmental IO model by exploiting abundant U.S. energy and environmental related statistics. However, studies pertaining to the sub-national environmental IO model are still rare. This review indicated that the conventional IO framework should be extended when it investigates the economic and environmental impact of different management options for EOL products and suggested that regional-specific environmental data that reflect technological differences among regions are ideal for constructing a regional environmental IO model. Therefore, this research will seek to create a new industry sector specified for CDW and carpet recycling in the regional IO model and to explore possible region-specific data sources for energy use and GHG emissions.

CHAPTER 3. THEORETICAL MODEL FOR GROWTH OF THE RECYCLING INDUSTRY

3.1. Conceptualization of Growth of the Recycling Industry

This purpose of this chapter is to present a theoretical model that explains how the recycling industry grows and how recycling facilities are spatially distributed. The theoretical model is built on an integrative framework that identifies key entities whose decisions are influential on the growth of the recycling industry and spatial patterns, and external factors that promote or restrain the growth of recycling industrial activities. The framework includes the following four key elements: institutional context, strategic company decisions, recycling technology, and regional market conditions and industry structure, displayed in Figure 1. These elements interact with one another and influence the growth of the recycling industry.

Institutional Context: The institutional context is an essential foundation upon which one can start to examine the growth of the recycling industry. One barrier to the growth of the recycling industry is the wide distribution of waste and recyclable materials across urban areas. The recycling of dispersed recyclable waste requires societal coordination of diversion activities as well as significant infrastructure investment and operating cost of collection. This condition leads to uncertainty which prevents the entry of new firms in the recycling industry. The basic institutional role clarifies and designates the entities responsible for diverting covered materials, investing in a collection infrastructure, and operating a recycling process. The creation of institutional rules, thus, can facilitate the development of a new social and economic foundation upon which a new recycling business can launch.

An institutional framework for recycling may be initiated by either the imposition of a legal obligation or voluntary social coordination. For example, some EOL products containing toxic substances are subject to mandatory recycling according to federal or state law. Municipalities can also establish their own mandatory recycling ordinances. Many municipalities in the San Francisco metropolitan area have passed local ordinances that require the diversion of some CDW materials. Alternatively, social coordination such as voluntary agreements and certification programs can create situations in which firms voluntarily participate in recycling even though it is not economically feasible in the short term. For example, the green building certification program provides building owners with incentives to become involved in the recycling of materials from their construction and demolition projects. Another notable example of self-regulation in recycling is the voluntary agreements within industry associations. In such cases, members of the associations initiate their own individual or collective recycling programs. Carpet recycling is a case of legally non-binding social coordination built upon the principle of extended producer responsibility (EPR).

Strategic Company Decisions: The strategic decisions of firms are critical determinants of the organizational reactions and the spatial distribution of recycling facilities. Thus, the conceptual model needs to account for firms as key decision makers by answering two questions: Why does a firm enter into recycling industrial activities in terms of collection and processing? And what organizational form will a firm take when it becomes involved in the recycling industry? These questions form a key dimension of the spatial growth pattern of the recycling industry. The decision to enter the recycling business can be influenced by various internal and external factors such as the ethical motivations of organizational decision makers, long-term managerial strategies, business competition in the green market, technological feasibility and cost savings, and new business opportunities.

After deciding to enter into recycling business, firms must decide the form of an organization that actually conducts recycling functions. This research suggests four possible types of organizational forms: outsourcing, consortium, in-house, and joint venture. The choice will differ according the transaction cost, the characteristics of EOL products, the technology of recycling, and conditions of the market. In addition, each organizational form may have different spatial implications. Consequently, this research predicts that the diverse spatial patterns of recycling systems manifested in the strategic choices of responsible firms.

Technology of Recycling: The frontier of recycling technology is an external factor associated with the strategic decisions of firms and the institutional settings. Technological and economic feasibility varies across the types of EOL products. For example, some EOL products suffer from degradation in the quality of recovered materials. Waste concrete is technically easy to recycle, but the performance of recovered aggregate, the products of waste concrete recycling, is typically inferior (e.g., in compressive strength) to that of virgin aggregate, which results in end use with lower economic value. By contrast, nylon 6 in the face fiber of waste carpet can be recycled without any loss of its desirable physical property, so it has a high economic value as a recycled material. In addition, some EOL products have multiple recycling options. For example, wood waste can be re-used through cleaning, denailing, and re-sizing and be recycled as the raw material of pulp or be burned as an alternative fuel. Company decisions are typically constrained by the technological and economic feasibility. In addition, each applied recycling technology has a different level of capital investment requirement and scale economy. Because the scale economy of recycling facilities is directly associated with the geographic scope of a recycling facility, the feasibility of recycling technology will be an important explanatory factor for the spatial pattern of recycling facilities.

Regional Market Conditions and Industry Structure: The regional market of virgin and recovered materials and the structure of industry structure are additional external influences on the spatial growth pattern of the recycling industry. Recovered materials are sold to local manufacturers as intermediate inputs or to consumers as final products. The provision of recovered material by a recycling facility may create new local or regional supply chains. Inversely, decisions regarding the location of a recycling facility must consider existing locational patterns of potential demand and supply links. In particular, in the case of EPR, a responsible firm more flexibly takes into account the supply chains and spatial patterns of recycling and existing manufacturing facilities.

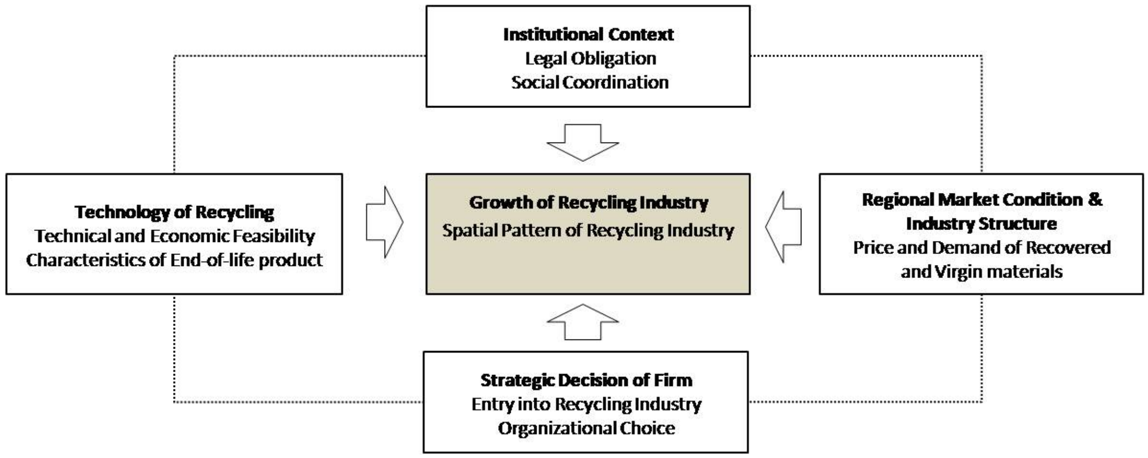

Figure 1: Schematic Map of the Growth of the Recycling Industry and Related Factors

By considering four elements, this research investigates the recycling systems of two groups of waste: 1) construction and demolition waste and 2) waste carpet. These cases illustrate two different recycling systems in terms of their institutional approaches, their technological complexity, the involved firm types, and their spatial dimension. They display the diverse recycling systems that exist institutionally, technologically economically, and spatially.

CDW recycling represents a localized recycling system. A local ordinance that mandates the diversion of CDW in construction, remodeling, and demolition projects is a primary institutional regulation for the development of a recycling system, and franchise waste management companies or private recyclers are the key entities that invest in sorting and processing equipment. That is, local regulations, franchise contracts, and physical characteristics related to CDW contribute to the growth of localized recycling systems. Waste carpet recycling, however, is a case of regional- or national-scale recycling systems. The primary institutional rule of carpet recycling is the industry-wide voluntary agreements among state and federal environmental protection agencies, major carpet manufacturing companies, and the industry association. In this approach, an original product manufacturer (OPM) plays a proactive role in managing EOL products and establishing a national- or region-wide recycling system. Two cases of recycling systems will be described in Sections 3.2 and 3.3.

Table 2: Comparison Between Waste Carpet and Construction and Demolition Waste Recycling

Feature	Waste Carpet Recycling	Construction and Demolition Waste
Institutional Approach	National-level private-public voluntary agreement(Carpet America Recovery Effort); local-level economic development approach (King County, WA, Linkup program)	Local ordinance of mandatory CDW diversion (San Francisco Metropolitan Area); local economic development approach (King County, WA, Linkup program)
Recycling System Governance (inter-firm connection)	Mix of two systems: 1) a closed-loop system within a vertically-integrated firm; 2) specialized processors and local collectors often subcontracted to vertically-integrated firms	Public and private transfer stations and private mixed CDW processing facility; inter-firm connection of small- and medium-size recycling firms by supply chain linkage
Industry Structure	Oligopolistic structure of an original product manufacturer: vertically- integrated firms operating at the regional or national level	Local-based competitive structure: a franchise waste management company and a small- and medium-size recycling company
Technological Complexity of Processing	Medium: process consisting of sorting, pelletizing, extruding, molding, and chemical processing; different applicable techniques.	Low: relatively simple process includes sorting, separating, crushing, grinding, denailing, and re-sizing
Demand Condition and Potential Market Outlet	Reclaimed nylon 6, carpet padding, plastic parts in auto, pallet, building material, and pellets	Wood: waste-to-energy conversion, reuse, engineering wood products, mulch Aggregate: sub-base in road construction, alternative daily cover in landfills, aggregate in concrete
Spatial Boundary of Industrial Activities	National- or regional-scale recycling	Local-scale recycling

3.1.1. Responsibility for End-of-life Products and an Institutional Approach

For several decades, the management of EOL products has been the responsibility of local governments in the U.S. Recently, however, the responsibility for several EOL products has shifted to the manufacturing industry. In light of this change, the institutional arrangements of the management of EOL products have diversified. This section examines two institutional approaches of responsibilities of local governments and manufacturers and explores possible organizational forms found in each institutional approach.

Responsibility of Local Governments

Historically, as one aspect of solid waste management in the U.S., recycling has been the task of local governments. In the late 1890s and early 1900s, public health problems associated with managing solid waste and sewage arose. Whereas the sewage issue was a relatively complex problem for local governments, requiring extensive capital investment on regional sewage infrastructures and regional-level cooperation, the solid waste issue was relatively easily resolved through local waste collection and dump services provided by local governments at that time (Louis, 2004). Local governments organized departments charged with cleaning the streets and disposing waste and adopted advanced sanitary engineering techniques into their local government practices. Subsequently, local government organizations formally institutionalized solid waste management, passing local ordinances regulating service areas, waste collection and disposal methods, financing methods, recycling programs, and law enforcement, and covering the types of products for collection and recycling.

In general, since local governments exclusively provide solid waste management services, the solid waste management is a local monopoly market. In other words, local recycling service for covered EOL products, a subset of local solid waste management, is mostly dominated by a monopolized service provider of recycling that can be either a public entity or a for-profit company. Although non-profit grassroots recycling organizations have been a part of local landscapes since the late 1960s, they have almost disappeared because of a lack of financial and human resources (Pellow et al., 1999; Lounsbury et al., 2003). Like the direct provision of solid waste management services by local governments, a common practice, the outsourcing of local solid waste management has also a long tradition (Warner and Bel, 2008). According to a survey of the Profile of Local Government Service Delivery Choices 2007, half of solid waste disposal services were delivered by for-profit companies or franchises (International City/City Management Association [ICMA], 2007).

An outsourcing contract is a form of privatization of a local public service. The rationale for privatization relates to cost savings by creating competition. The local government can create a bidding market in which private waste management companies compete against one another. Market competition is expected to allow a local government to select the most competitive service provider, which are often consolidated large private waste management companies. Such companies may provide a cost-effective waste management solution through operations of large-scale collection and landfill infrastructure and investments in hauling and processing equipment. Thus, private companies of waste management with a franchise contract play a key role in the growth of the recycling industry under the responsibility of local governments.

Extended Producer Responsibility

Recently, as the role of manufacturer for EOL products has been emphasized, EPR has emerged as an alternative institutional rule (Fishbein, 2000; Wall, 2006). EPR aims to shift the responsibility for managing EOL products from final users to manufacturers. Fundamentally, it intends to transform industrial production practices by incorporating the element of life-cycle product management into product design (Lifset, 1993). The principle of EPR has been adopted in Asia, Europe, and North America for diverse types of products such as packaging, carpeting, automobiles, and electronic products.

The participation of industry in the management of EOL products may be motivated by environmental stewardship, a strategic decision for long-term profit maximization, legal obligation, or possibly the nexus of moral and profit-seeking motivations (Lyon, and Maxwell, 2008). Several studies have examined the economic justification of firms that strategically choose to voluntarily become involved in environmental management such as the establishment of closed-loop systems, waste minimization, and design for the environment (Esty and Porter, 1998). The management of EOL products and design for the environment can enhance the value of a firm if it actually capitalizes on potential benefits such as saving on the use of materials that would otherwise go unnoticed, lowering risk associated with unanticipated contamination and compliance with stricter regulations, and improving competitive advantage through product differentiation (Reinhardt, 1999). However, Esty and Porter

(1998) also contended that the potential for capitalizing on environmental management can be thwarted by excessive costs and fragmented policies and regulations.

The principle of EPR has been institutionalized in two ways, through legislation and voluntary agreement. Typically, in both cases, a regulatory agency plays a proactive role in the adoption of EPR. The regulatory agency may lead to enacting legislation based on the principle of EPR. This direct regulation through legislation has been observed for several products such as automobiles, batteries, and electronics (Toffel, 2003). Voluntary adoption of the EPR policy by the industry may also be influenced by regulatory agencies. If an industry anticipates that stricter regulations will likely be imposed in the near future, it may be motivated to participate in a voluntary agreement scheme or to develop its own environmental program in order to pre-empt the threat of regulation.

For implementation of the principle of EPR, several environmental political programs with different features such as physical versus financial responsibility and collective versus individual responsibility have been initiated (Toffel et al., 2008). Each program along with its unique features has advantages and disadvantages, and those are associated with the strategic decision by a firm with regard to its organizational form.

The principle of EPR can operate either collectively or individually. In the collective scheme, firms within the same industry organize a consortium that commits to implementing collection and recycling, and each member of the consortium is charged a portion for up-front investment and operational costs. The advantage of collective responsibility is that it can reduce the cost of operation when it achieves economies of scales in the recycling facility. However, the disadvantage is that the incentive for modifying a product design in more environmentally benign way is weakened as a result of the free-rider problem (Lindquist and Lifset, 2003) because the benefit of re-design for disassembly and recycling does not exclusively accrue to an innovative firm in the collective scheme. A possible solution to the free-rider problem is a differentiated fee for a brand name. A notable example of a differentiated fee is the green dot system applied to packaging waste, for which a responsible firm pays a fee that varies according to the weight and the material used for packaging. Another disadvantage of collective responsibility is the cost of the coordination. A fair, efficient mechanism for assigning financial responsibility to individual participants in a consortium is necessary for successful operation.

The individual responsibility scheme, which relies on the commitment of a single firm, has the advantage of incentivizing product redesign and requiring no coordination costs. However, under a voluntary agreement without a policing mechanism, the attainment of intended objectives cannot be guaranteed (Fishbein, 2000). Thus, the strategic choice of whether a firm creates an individual recycling system or joins a consortium depends on the capacity of participating firms, the recyclability of products, and the market outlook for recycled materials.

The physical responsibility model entails the direct involvement of the OPMs in collecting, transporting, and recycling EOL products. It is a form of "in-house" recycling. A firm that takes physical responsibility can benefit from learning and innovation opportunities (Pagell et al., 2007). That is, the firm may learn about environmental innovation in product redesign by directly dismantling and recycling its own EOL products. Alternatively, the financial responsibility model entails the handling of actual jobs related to collection and recycling by a third-party recycler, and the OPMs pay for cost of collection and recycling. It is outsourcing recycling. When neither a closed-loop system nor recycling is immediately beneficial to the OPMs, they may prefer the financial responsibility model.

An EPR policy can be implemented by diverse environmental programs. An OPM can take several organizational forms, including consortium, in-house, and outsourced recycling. Since each has advantages and disadvantages, OPMs choose one appropriate organizational form. This choice will be thoroughly examined in the next section.

3.1.2. Original Product Manufacturer's Decision on the Vertical Integration of the Recycling Function

This section raises the following core question: Why does a firm choose either an outsourcing recycling option or an in-house recycling option? To provide insights into answering this question, this section presents an illustrative model adapted from Scott (1983, 1986) in the lens of transaction cost theory that accounts for factors that influence an OPM's decision about its organizational form.

Let us assume two products, a and b. Product a is an intermediate input of the production process of b. The cost of the production of a and b is denoted $c(a)$ and $c(b)$. The cost curve is assumed to be a U-shape. First, if we consider the cost structure of vertical integration in which a firm manufactures both products a and b in their own facility, the total production cost of this facility is the sum of the individual production cost of a and b as well as the cost associated with vertical integration. Two types of cost items are conceivable for vertical integration. One is obviously the cost of coordinating two production processes. The other is the effect of the economies of scope, which is actual cost savings. One production process may complement the other. The cost related to vertical integration is denoted $v(a,b)$. The cost of vertical integration is assumed to decrease as the output level increases because the cost of coordination is relatively fixed regardless of the output level, and the effect of the economies of scope becomes stronger as the output level rises. Thus, the cost curve of $v(a,b)$ is a downward slope, shown in the first graph of Figure 2 ($v(a,b)^{'} < 0$). If the effect of the economies of scope exceeds the cost of coordination, a firm can capitalize on the benefit of vertical integration. The total production cost of a vertically integrated firm will be denoted $c(a) + c(b) + v(a,b)$.

In a case of the production cost of vertical disintegration, a firm manufactures product b and purchases intermediate input, product a, in the external market at price p_a. The price of p_a is assumed to be constant regardless of the output level of X_b. An external market transaction incurs additional costs such as those related to searching and contracting and risk associated with executing contracts and ensuring the quantity and the quality of input products. Costs pertaining to the market transaction is denoted $d(a)$. When the size of a supply contract increases, the contracting cost per unit will decrease. The cost related to risk will decrease when there is long-term contract. The cost of disintegration decreases as the output level rises ($d(a)^{'} < 0$). The total cost of vertical disintegration will be denoted $c(b) + p_a + d(a)$, shown in the second graph of Figure 2.

The decision by a firm about whether to manufacture or purchase will depend on the cost curves of vertical integration and disintegration. The relationship between the cost curves in both cases is illustrated in the third graph of Figure 2. In an area in which the cost of vertical integration is lower, a firm is willing to combine two production processes in its operation. In an opposite type of area, a firm is willing to abandon the production process of the intermediate input and purchase input product from the market.

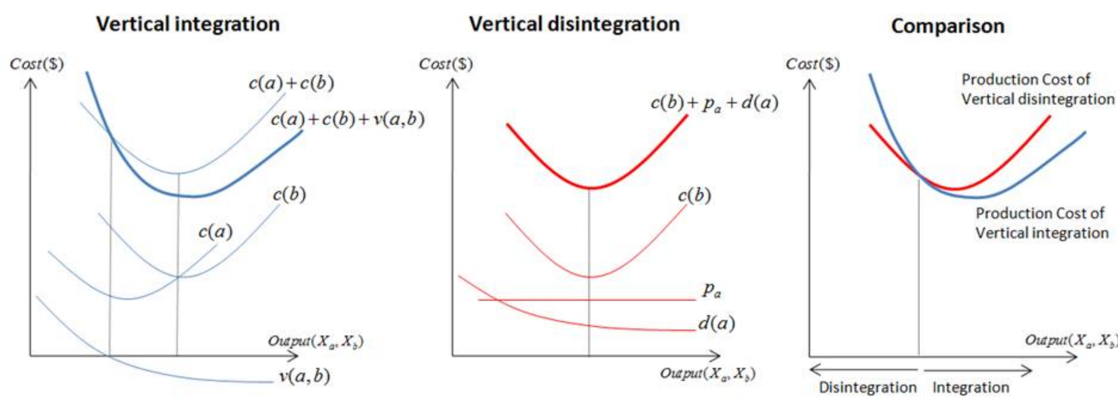

Figure 2: Comparison of the production costs of vertical integration and disintegration

Source: Adapted from Scott (1983, 1986)

Applying this illustrative model in the context of the recycling industry, this research first considers the complementary relationship between the original production process and the recycling process, a key factor influencing the extent to which an OPM will choose to integrate its recycling functions. If waste collection is the responsibility of a local government, a franchise waste collection company can obtain benefits by combining the collection service, an original production activity, with a recycling service. When a waste recycling program is run by local governments, the costs of recycling combined with existing collection systems will be lower than the costs of separate collection and recycling. In addition, the franchise collection company earns extra profit from selling recovered materials made from collected waste. Thus, the franchise waste collection company is highly likely to invest in its own recycling facility connected to a waste collection system.

Under EPR policy, the extent of technological complementary depends on the types of products and manufacturing processes. For example, if an OPM is a simple assembler of various components, a synergistic relationship in the assembly OPM may be extremely difficult to form. This condition can result in outsourcing the recycling process. However, if the recycling of the EOL products of an OPM has a technological feedback effect on its original production process or if recycling involves specialized knowledge, an in-house recycling system is preferable. In an ideal closed-loop system, product design will encompass the disassembly and recycling stages and knowledge of recycling will influence the original production process, leading to an integrated organizational form and production system.

Second, the notion of the asset specificity of the recycling process will also influence the decision pertaining to vertical integration or disintegration. When a product or material reclaimed from the recycling process is a highly customized intermediate input, a vertically- integrated process is more likely. When recycled materials can be used for multiple purposes, a vertically-disintegrated form of organization is more likely.

Finally, under market uncertainty, vertical integration is a more likely means of avoiding risk associated with external contracts in general. The recycled material market is highly uncertain in terms of price fluctuations and unstable supplies of waste materials. However, the effect of market uncertainty of recycled materials will differ according to the strategy of OPMs. If a firm intends to establish a fully closed-loop system in which recycled materials are re-introduced in the original production process, the uncertainty will motivate a vertically- integrated organization. Alternatively, if a firm does not anticipate complementary benefits from the integration of the recycling function or if OPMs sell recycled material to the market, market uncertainty leads to vertical disintegration.

3.1.3. Industry Organization and Spatial Linkages of the Recycling Industry

This section discusses the features of the organizational forms that firms may take for recycling, and then draws the possible spatial implication of recycling systems under two institutional contexts. When the responsibility for recycling lies with local governments, the recycling system is a traditional localized recycling system; however, the EPR policy adds new dynamics to the organizational forms and the spatial patterns of recycling systems.

Local Government Responsibility and Localized Recycling Systems

As discussed in a previous section, the responsibility of a local government for solid waste management creates a locally monopolistic recycling market. The mandatory recycling program of local governments requires new investment in recycling collection systems and processing facilities. Recycling bins and roll-off containers for source separation need to be provided to those who dispose waste, and a publically or privately owned recycling facility has to be built. Such a program serves a local area with an administrative boundary.

The question is whether or not a franchise for-profit company is willing to invest in the recycling infrastructure. Lounsbury et al. (2003) noted that major companies of solid waste management, which are vertically integrated, thought that they could earn additional revenue through recycling from waste streams that were already collected. Since they operated collection systems and owned large-scale landfills, the addition of a recycling function to a vertical integrated company may have been viewed as an opportunity to maximize the value of their collected waste. The location pattern of recycling under the control of a local government is relatively straightforward. The recycling function falls in the middle of the flow from waste collection to disposal. Thus, the recycling facility is more likely to co-locate with local transfer stations or landfills.

Extended Producer Responsibility and Diverse Location Patterns

Adoption of the EPR policy by industry entails making strategic decisions pertaining to the organizational form categorized into contract outsourcing, joint ventures with a recycling expert, industry consortiums, and in-house vertical integration (Toffel, 2003; Pagell et al., 2007). Each organizational form has different features.

Outsourcing is a relatively easy, cost-effective option in the short term. If OPMs are forced to participate in recycling due to legal obligations and if recycling is economically infeasible in the short run, OPMs are more likely to choose an outsourcing option. Thus, the outsourcing option offers an opportunity for subcontracted small businesses that can operate at the metropolitan or regional scale for collection and processing.

An industry consortium is an alternative type of outsourcing. Under fragmented regulatory conditions and economic infeasibility, OPMs may prefer collective responsibility. OPMs participating in the consortium can either establish a shared collection system and processing facility, or collectively contract with local independent recyclers. In both outsourcing and consortium options, the recycling system is more likely to be disconnected from the manufacturing system of original products, and as a result, an open-loop recycling system may occur. The forward linkage of recycled material use is not limited by the operation of OPMs either functionally or spatially.

In-house recycling is a relatively expensive option because of the up-front capital investment for collecting and processing EOL products. In-house recycling will become more economically feasible when recycling systems become standardized and operationalized on a large scale and when they are able to reclaim valuable recycled materials that can be re-introduced into the original manufacturing process. The in-house option has some potential long-term advantages. For one, OPMs may acquire new knowledge through the learning process of direct involvement in terms of economizing material use, improving the performance of recycled material, refining product design cost-effectively and in an environmentally benign way, and creating new value from recovered material. Moreover, through vertical integration, OPMs can avoid risks such as uncertainties incurred of supply chain. The OPM with an in-house recycling system either develops its own reverse collection system or relies on a municipality's collection systems and local hauling contractors. If OPMs already have established logistics for the distribution of their production, including regional warehousing and wholesaling, these warehouses may be utilized in reverse logistics.

Finally, the joint venture is a flexible option for OPMs that reduces the burden and the risk of up-front capital investment and shares the knowledge of recycling with partners who specialize in the recovery and recycling process. The joint venture may conduct scientific research to improve the recyclability of material and the functional performance of recycled material and recycled-content products.

Each organizational option carries unique implications for the location pattern and spatial linkages of recycling. With regard to outsourcing and industry consortium options, EOL product management is least likely to be integrated into the product design and original manufacturing process. The location of reverse logistics and the processing facility may not be a high priority in the strategic decision making of OPMs. Thus, the primary location factor for recycling facilities is that of minimizing collection costs, which is beneficial for independent subcontracted recyclers subcontracted or recycling facilities commissioned by a consortium. Therefore, the location of the recycling facility is oriented to an urbanized area where the majority of consumption and disposal take place.

In the joint venture and in-house options, the forward and backward linkage in the integration of the production system is an important factor. OPMs need to take into consideration spatial relationships among recycling and other branch facilities in which recovered materials will be consumed. Consequently, the vertically-integrated recycling facility is more likely to locate sufficiently close to other facilities of the OPMs. As an alternative form, OPMs can operate a regional center charged with collection and pre-processing such as dismantling and shredding. This process is responsible for separating valuable parts from the rest of EOL products and shipping only recyclable materials to centralized recycling facilities (Toffel, 2003).

In summary, the strategic choices of OPMs create new dynamics in the spatial patterns and linkages of recycling activities. In addition, by adoption EPR policy, traditional local-based recycling activities can be extended to the regional or national scale. The actual extent of the geographic scale may be contingent upon product types, applicable technology, and existing industry structure. The types of industrial organization and associated spatial implications are summarized in Table 3. In the next section, this theoretical framework will apply to two cases of CDW and carpet recycling.

Table 3: Industrial Organization and Spatial Pattern of Recycling Systems

Outsourcing • Cost-effectiveness for OPMs in the short-run • Delinking from design for the environment • Vertical disintegration • Small business opportunities • Urban-oriented location	Consortium • Collective responsibility • Economies of scale • Vertical disintegration • Cost of coordination • Serving the area that legislation designated • Urban-oriented location
In-House • Expensive in the short-run • Up-front capital investment • Possible to standardize • Vertical integration • Use of recycling materials in original manufacturing process • Design for the environment • Local and regional collection center • Located near OPMs, but geographically loosely confined	Joint Venture • Technical innovation • Flexible operation • Avoidance of risks • Semi-integration • Communication with OPMs • Located near OPMs

3.2. Case of the Recycling of Construction and Demolition Waste

The San Francisco metropolitan area is a notable case of CDW recycling, representing a public-driven localized recycling system combined with for-profit waste management companies. This section describes the development of localized CDW recycling systems by reviewing state and local policies and private sector involvement.

3.2.1. Local Ordinance and Recycling Program for Construction and Demolition Waste

State regulations and the establishment of local ordinances were major drivers of the development of local CDW recycling systems in California. The milestone of state legislation for recycling was the Integrated Waste Management Act (Assembly Bill [AB] 939) of 1989. At that time, concerns about the shortage of permitted landfills were growing, anticipating that existing landfill capacity would be exhausted by 2000 (Report of the Little Hoover Commission, 1989). Corresponding to this concern, the Integrated Waste Management Act required local governments to achieve 50% diversion of their municipal solid waste by the year 2000.[3] To comply with the state's mandate, each local government developed its own recycling program and considered regulating CDW, which could potentially increase the diversion rate.

Local governments found several advantages of targeting CDW in the mandatory diversion plan. First, CDW comprised a large portion of municipal solid waste. In the economic upswing of the 1990s, California experienced a building construction boom that resulted in an increased volume of CDW. A mandatory recycling ordinance would

[3] The history of the California Environmental Protection Agency, website accessed March 2012 at http://www.calepa.ca.gov/about/history01/ciwmb.htm

significantly contribute to the attainment of the diversion goal. Another advantage of targeting CDWs is that diversion activities for CDWs are more controllable by the local government than those for other types of waste because the government can combine mandatory diversion regulation with the building construction and demolition permit process. A permit is issued only to those who comply with a local diversion rate set by the local ordinance. (California Integrated Waste Management Board, 2003).

By considering these merits, most local governments in the San Francisco metropolitan area established local ordinances for the mandatory diversion of CDW and proactively initiated CDW recycling programs. Local ordinances related to the diversion requirement are listed in Table 4. This requirement is linked to the building permit process. For example, in the City and County of San Francisco, to obtain a demolition permit at the beginning of a project, the demolition project contractor should complete a Demolition Debris Recovery Plan. The Demolition Debris Recovery Plan indicates how the demolition project can achieve the minimum 65% diversion rate. After completing an actual demolition project, a project contractor must also submit a final report that certifies that the required amount of CDW has actually been transported to recovery facilities. To ensure compliance with their diversion plan, several local governments such as the City of San Jose have employed a deposit program in which each contractor of a construction and demolition project pays a deposit when applying for a project permit. The deposit is refunded after a contractor submits necessary supporting documents once a project is complete.

Table 4: Selective Cases of CDW Recycling Ordinance in the San Francisco Metropolitan Area

City	Diversion Requirement	Threshold
City and County of San Francisco	65% of waste generated	All construction, demolition, and remodeling projects
County of Contra Costa (unincorporated area)	50% of waste generated	Project > 5,000 sq ft.
County of San Mateo (unincorporated area)	100% of inert solid and 50% of remaining waste generated	Demolition > $5,000 Construction project > $250,000 Or >2,000 sq. ft. Remodel
County of Alameda (unincorporated areas)	75% of inert solids50% of remaining waste generated	All demolition projects, residential projects >1,000 sq. ft, Commercial projects >3,000 sq ft.
City of Alameda	50% of waste generated	Projects valued at $100,000 or more
City of Berkeley	100% of concrete and asphalt, 100% of land clearing waste, and 50% of remaining waste generated (Applicants shall make salvageable materials available for reuse prior to demolition)	All new construction renovation projects valued at $100,000 or greater; all demolition projects over $3,000 valuation.
City of Fremont	100% of concrete and asphalt 50% of remaining waste generated	Construction and renovation projects valued $300,000 or greater (residential, commercial and civic); all demolition projects
City of Oakland	100% asphalt and concrete 65% of remaining waste generated	All new construction, All demolition projects, commercial projects valued at $50,000 or more
City of Union City	50% of waste generate	Construction and demolition projects valued at $100,000 or more; residential remodels increasing square footage by 50% or more

Source: SF Environment, Department of the City and County of San Francisco accessed September, 2012 at http://sfenvironment.org/article/construction-amp-demolition/construction-and-demolition-resources; Central Contra Costa Solid Waste Authority accessed September 2012 at http://www.wastediversion.org/app_pages/view/31; RecycleWorks: A Program of San Mateo County accessed September, 2012 at http://www.recycleworks.org/con_dem/index.html; Alameda County Waste Management Authority accessed March, 2012 at http://www.stopwaste.org/docs/ordinance_matrix_gb.pdf

3.2.2. Construction and Demolition Waste Recycling Technology and Material Flow

With the development of local institutional rules, the physical infrastructure and associated businesses for diversion, collection, and processing are key aspects to the implementation of a CDW recycling program. Three types of businesses relate to the flow of CDW recycling, shown in Figure 3. Diversion is the initial stage of sustainable CDW management. The extent of diversion in construction and construction projects depends on the effort of construction or demolition contractors. During a project, a contractor can conduct diverting activities, including both on-site recycling and source separation, and apply the deconstruction technique. From an environmental standpoint and from a job creation potential, deconstruction is preferable, so several for-profit demolition companies and non-profit organizations provide deconstruction services in the San Francisco metropolitan area. Its application, however, is relatively limited.

Another key element in the implementation of a CDW recycling program is the collection system. Local governments in the San Francisco metropolitan area signed exclusive franchise contracts with waste haulers or had non-exclusive register systems. Since the generation of CDW is sporadic, the registered or franchise haulers lease roll-off bins to the construction or demolition contractors. When a bin is loaded, the hauling company transports it to designated sites.

Because a large amount of diverted CDW is hauled without source separation to recycling facilities, the mixed CDW processing facility plays an important role in building a local recycling system. This facility is a combined process of manual labor, heavy equipment vehicles (i.e., end-front loaders and excavators), and mechanical equipment (i.e., screens, horizontal conveyers, wood grinders, magnet separators, float tanks, and concrete crushers). The recycling rate differs among registered faculties. According to a report of the Alameda County Waste Management Authority (ACWMA) in 2011[4], the recycling rates of mixed CDW processing facilities in the San Francisco metropolitan area ranged from 67% to 98%, including the use of alternative daily cover (ADC) in landfills.

The local government report (North Central Texas Council of Government, 2007) examined the economic feasibility of a mixed CDW processing facility. According to their estimates, revenue from selling recovered materials (i.e., crushed aggregate, grinded wood, sorted metal, baled cardboard, grinded gypsum, and salvaged products) covers 25.3% of the total in the worst case and 49.6% in the best case, and the mixed CDW processing facility needs to charge a fee of $19 to $25 per ton on incoming mixed CDW for profitable operations. If a fee of a mixed CDW processing facility is lower than the tipping fee of the landfills, a mixed CDW processing facility may be competitive in the CDW management market. Since the San Francisco metropolitan area is one of the highest tipping fees[5] among all of the landfills in the U.S., a mixed CDW recycling facility may take advantage of it in the CDW management market. This report also investigated job creation in the mixed CDW processing facility and found that a facility that processes 70 tons of CDW per hour can hire about 30 employees, including

---

[4] The information about diversion and recycling rate of mixed CDW recycling facilities is available in the website of ACWMA. http://www.stopwaste.org/home/index.asp?page=292

[5] The rate of the tipping fee differs according to the types of waste. For example, the tipping fee of mixed CDW is $121 and that of separated wood waste such as lumber and pallets is $60 in Waste Management, Inc. Approximately, the tipping fee for mixed CDW ranged from $65 to $125 in the San Francisco metropolitan area.

supervisors, equipment operators, mechanics, and manual laborers, the latter of which makes up two-thirds of the employees.

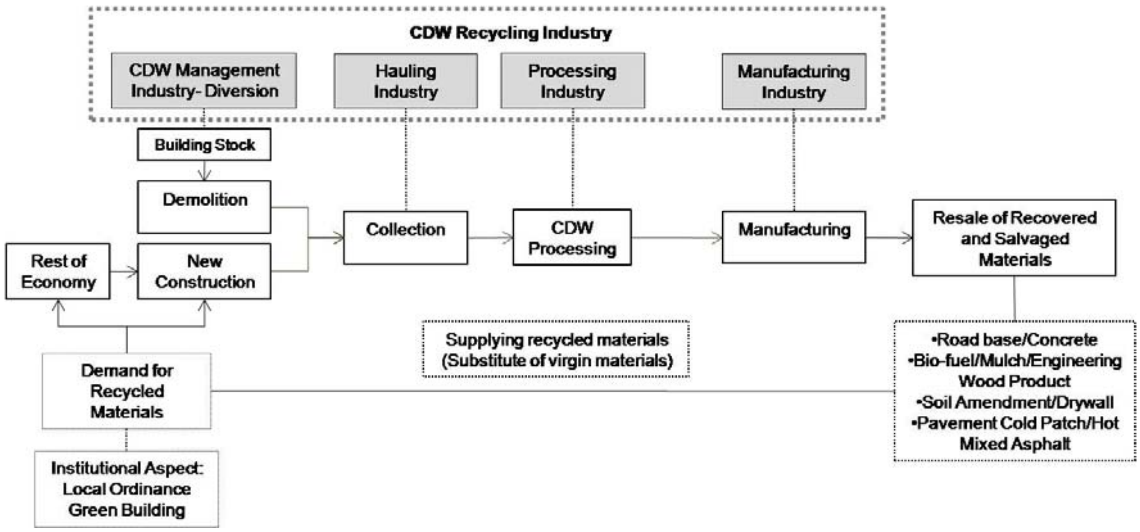

Figure 3: Construction and Demolition Waste Recycling Flow

3.2.3. Development of a Localized Construction and Demolition Waste Recycling System

CDW recycling in the San Francisco metropolitan area is a localized system rooted on the nexus of public-driven initiatives and the involvement of private companies. After local ordinances governing CDW diversion in most municipalities were established and the diverted volumes of CDW were expected to increase, the local governments needed to ensure the sufficient local processing capacity of facilities that properly handle diverted CDW. To expand local processing capacity, they either directly invested in building processing facilities or created incentives for private companies to invest in their facilities. Therefore, it was the support of the local government and investment by the private sector that fostered the development of local CDW recycling systems in the San Francisco metropolitan area.

In their effort to encourage private companies to invest in processing facilities, local governments can offer several policy options. First, they can financially support private sector firms. One example of such direct financial support is the partnership between the Alameda County Waste Management Authority (ACWMA) and Waste Management, Inc. In 2000, Waste Management, Inc., one of the largest waste management companies in the U.S., decided to build a $2.6 million material recovery facility at the Davis Street Transfer Station in the City of San Leandro, California, and the ACWMA agreed to provide underwriting assistance to the company for the construction of this facility.[6] The ACWMA funded the material recovery facility to recycle at a rate of $15 per ton from 2002 to 2007. During this period, the average amount of eligible diverted material was 31,000 tons per year.[7]

Another example of direct support was grants bestowed by the Construction and Demolition Recycling Infrastructure Grant Program of the City of San Jose in 1999/2000 and 2000/2001. One company to receive this grant was Zanker Road Resource Management and Waste Management, Inc. With the grant, the company, which had sorted and recycled CDW since 1988, expanded its sorting and processing facility by installing an organic removal and screening system; then another $100,000 grant[8] was awarded to Waste Management, Inc., which built a 20,000 square-foot

---

[6] *Recycling Today* November 21, 2000, accessed in March 2012 at http://www.recyclingtoday.com/Article.aspx?article_id=16338

[7] Alameda County Waste Management Authority Program/Planning Committee, Minutes, December 11, 2007.

[8] The City of San Jose Website, accessed in March 2012 at http://www.sanjoseca.gov/clerk/6_5_01docs/2.4rev.htm

roofing recycling facility in the Guadalupe Landfill and purchased a mechanical and hand sorting line with a 200 ton per day processing capacity at a cost of $600,000. [9]

Another option of a policy support is exemption from state and local taxes related to the landfill industry. For example, the City of San Jose enacted the Disposal Facility Tax in 1992. In lieu of a business tax based on the number of employees, the operators of landfills located in the city were required to pay a tax based on the number of tons disposed in landfills at a rate of $13 per ton. However, materials that are salvaged and recycled are exempted from the tax on the first 33,500 tons of waste.[10] In addition, solid waste facilities in California are required to pay the Integrated Waste Management Fee at a rate of $1.40 per ton.[11] This fee can also be refunded as much as the amounts of accepted waste are reused or recycled. These measures provide financial incentives that promote investment by solid waste facilities in recycling processes.

Also utilized as a policy option is the approval system, in which only approved facilities can accept diverted CDW.[12] If a facility is neither qualified nor approved to recycle CDW, it may lose large portions of incoming waste materials. Thus, in order to be approved, private companies have to install processing equipment. Many municipalities in the San Francisco metropolitan area have register systems, certification, or approval systems.

Because of direct financial support, tax exemption laws, and approval requirements, several CDW processing facilities have been established, and they are

---

[9] *Biocycle*, March 2002, accessed March 2012 atwww.p2pays.org/ref/44/43183.pdf
[10] Integrated Waste Management Zero Waste Strategic Plan Development of the City of San Jose, Appendix C, accessed in March 19, 2012 at www.sjrecycles.org/zerowaste-stratplan.asp
[11] California State Board of Equalization, accessed in March 19, 2012 at http://www.boe.ca.gov/sptaxprog/ciwmb_solidwood_waste.htm
[12] The City and County of San Francisco, accessed in June 12, 2012 at http://greencitiescalifornia.org/assets/waste/SF_c-d_Brochure%20.pdf

currently operating in the San Francisco metropolitan area listed in Table 5. These facilities can be categorized into three groups: public processing facilities, private processing facilities operated by franchise companies that provide integrated waste management services, and private processing facilities owned by independent recyclers. The portion of direct involvement of the public sector is relatively small. For example, the public facility operating at the City of Berkeley Transfer Station received only 242 tons of CDW.

Unlike public facilities, private companies with franchise collection service played a pivotal role in CDW recycling. Major waste management companies such as Waste Management, Inc. and Allied Waste provided integrated waste management services from collection, single- and mixed-stream waste recycling, composting, and energy recovery to disposal. Since their CDW recycling facilities are a segment of the integrated waste management system, they co-located with composting, other material recovery facilities, and landfills. The advantage of co-location of related facilities is the complementary use of processed material. For example, grounded wood from CDW processing can be sent to an in-house composting facility, and non-recyclable residuals from CDW processing such as painted wood and dirt are used as alternative daily cover in their own landfills. Thus, we can conclude that CDW processing facilities operated by franchise waste management companies are more likely to co-locate with transfer stations within a dense urban area or landfills on the outskirts of an urbanized area, as displayed in Figure 4.

Table 5: Processing Facilities in the San Francisco Metropolitan Area

Name	Tons received	Diversion rate	Co-location	Note
City of Berkeley Transfer Station	242	88%	Transfer station	Publicly operated transfer station contracted with the Urban Ore for sales of reusable items
Commercial Waste and Recycling	18,062	90%	-	Independent CDW recycler handling medium volume
Davis Street Transfer State	1,836	69%	Transfer station	Transfer station owned by the Waste Management, Inc., a waste collection franchise Resource recovery complex of a single stream recovery, food and organic waste composting, and material recovery facility
Newby Island Sanitary Landfill (Newby Island Resource Recovery Park)	23,092	68%	Landfill	A subsidiary of Allied Waste (Republic Service), a waste collection franchise in Newby Island Resource Recovery Park: a gas-to-energy operation, a composting facility, a CDW recycling facility, a disposal operation
Pleasanton Garbage Service Transfer Station	4,602	72%	Transfer station	Recycling facility operated by the Pleasanton Garbage Service, a waste collection franchise
Recology-SF Recycling iMRF	4,890	75%	Transfer station	Operated by SF Recycling & Disposal, a subsidiary of Recology, a waste collection franchise
SRDC	4,678	98%	-	Independent CDW recycler handling a small volume
Zanker Materials Processing facility	256,296	67%	Landfill	Operated by Zanker Road Resource Management, a recycling, composting, and landfill company
Marin Resource Recovery Center	187,385	73%	Transfer station	Operated by Marin Sanitary Service, a waste collection franchise
Shoreway Environmental Center	-	-	Transfer station	Owned by the South Bay Waste Management Authority, a joint powers authority with twelve member agencies; operated by South Bay Recycling (renovated and opened in September 2011)

Source: Stopwaste.org, diversion/recycling rates for local mixed C&D processing facilities (November 2011); Marin County Hazardous and Solid Waste Management Joint Powers Authority, construction and demolition debris certified facility list (January 2012)

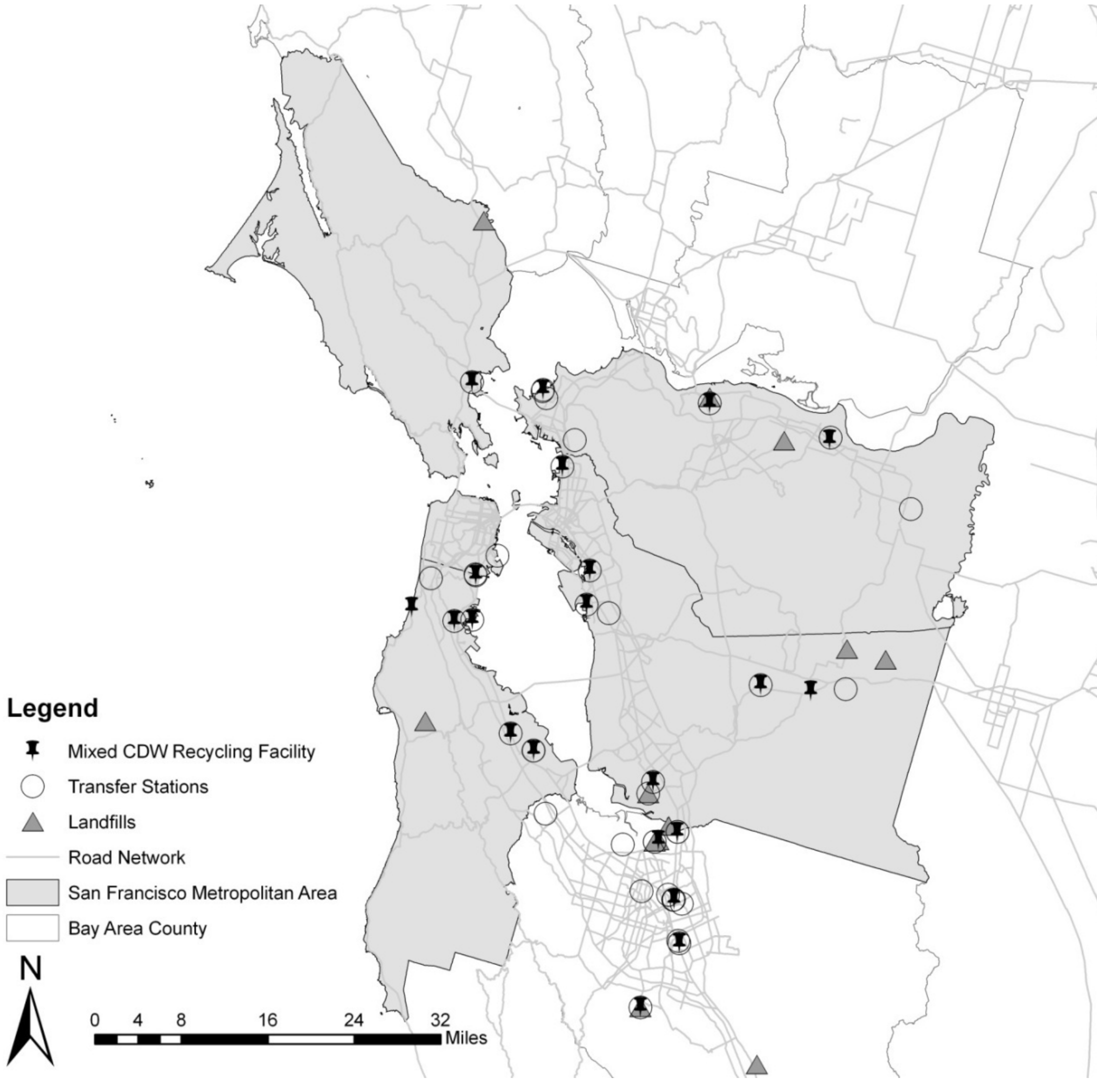

Figure 4: Location of CDW Recycling Facilities Transfer Stations, and Landfills in the San Francisco Metropolitan Area

Source: Solid Waste Information System, California's Department of Resources Recycling and Recovery (CalRecycle)

3.3. Case of the Recycling of Waste Carpet

In contrast to the localized recycling system of CDW, waste carpet recycling represents the case of regional- or national-scale recycling systems through public-private joint efforts based on voluntary agreements to adopt the principle of the EPR.

Waste carpet has a substantial potential for recycling in terms of quantity, material property, and technical feasibility. This section illustrates how diversified recycling systems for waste carpet have been established. First, it examines three external factors that influence the growth of carpet recycling systems such as the industry structure of the carpet manufacturing sector, the technology of recycling, and the voluntary agreement of carpet stewardship, and then illustrates possible organizational forms and spatial patterns of carpet recycling with cases.

3.3.1. Agglomeration of Carpet Manufacturing

The existing organizational structure of a carpet manufacturing company and its spatial distribution is a key factor that explains the growth of carpet recycling systems. Historically, the carpet manufacturing industry has been spatially clustered in northern Georgia around the City of Dalton, with the technological invention of the tufting process (Fuellhart, 1999). Because the size of the tufting machine was large, the components of the tufting machine had to be assembled in the manufacturing facility during installation. Thus, the proximity to carpet machinery manufacturers was a key determinant of the location of the carpet manufacturing industry (Walter and Wheeler, 1984). The invention of the tufting machine, combined with easy access to a supply of synthetic fiber,[13] has given northern Georgia the competitive edge.

Another significant feature of the carpet manufacturing industry is its highly oligopolistic structure and the vertical integration of major carpet manufacturers. The

---

[13] The synthetic fibers such as nylon and polypropylene were introduced in the 1950s. The use of synthetic fiber dramatically facilitated the growth of carpet industry by offering a durable, lower-price carpet products. A history of the U.S. carpet industry, accessed in July 25, 2012, retrieved from http://eh.net/encyclopedia/article/patton.carpet

carpet manufacturing industry experienced rapid growth in the 1960s and the 1970s as the price of carpet declined and carpet consumption per capita increased. During this upswing, small-size carpet manufacturers entered the market. However, in the economic recession of the 1980s, carpet consumption declined, and large carpet manufacturers engaged in mergers with or acquisition of small manufacturers as well as material suppliers. Consequently, vertical integration increased. The carpet manufacturers purchased yarn spinning, dyeing, and fiber production facilities, and established their own regional distribution centers. According to the 2007 economic census, the top four carpet manufacturers account for 63.4% of the total value of shipments.[14]

3.3.2. Waste Carpet Recycling Technology and Material Flow

The carpet manufactured in the U.S. consists mostly of synthetic petroleum-based materials such as nylon 6, nylon 6.6, polypropylene, and polyester (Wang et al., 2003). These synthetic materials in carpet products can be reused, recycled, and used in waste-to-energy conversion. This analysis focuses on recycling, the dominant activity in the diversion process. Currently 80% of diverted waste carpet undergoes recycling processes (Carpet America Recovery Effort, 2010). Although carpet recycling technology was reviewed in a literature section, this section revisits this topic in order to provide insights into the types of recycling businesses and spatial patterns associated with each applied technology.

Recycling techniques can be categorized into primary and secondary approaches. In the primary approach, the material in waste carpet is recovered in its original form

[14] U.S. Economic Census 2007, Website accessed on June 30, 2011 at http://factfinder.census.gov/servlet/IBQTable?_bm=y&-ds_name=EC0731SR12&-NAICS2007=31411&-_lang=en

with the same function. Nylon 6, used in face fiber, has desirable material properties for recycling (Segars et al.2003). Through a chemical and mechanical process, nylon 6 in waste carpet can be converted into recycled caprolactam, a feedstock for making recycled nylon 6. Purified caprolactam obtained from nylon 6 in waste carpet is comparable to virgin caprolactum (Wang et al., 2003). Recycling of nylon 6, potentially used in recycled-content carpet products, is a pure closed-loop system. In the secondary approach, the synthetic fibers in waste carpet are recycled as plastic materials, which have been used in a number of applications (Guidry, 2008; Subbiah, 2008). Face fibers such as nylon 6 and nylon 6.6 are reclaimed and become input materials such as carpet padding product and various plastic-molded products. It is an open-loop option.

These carpet recycling flows and associated businesses are exhibited in Figure 5. Waste carpet recycling typically goes through four steps: collection, sorting, processing, and manufacturing. In collection, waste carpet is separated from other municipal solid waste or construction and demolition waste. Because waste carpet contaminated by water or other debris is difficult to recycle, it must be kept dry and clean during the separation process. The growing replacement of carpet tile in commercial uses will facilitate the separation and collection steps in recycling (Lave et al., 1998). In the sorting step, sorting companies separate waste carpet according to color or material type by mechanically cutting off face fiber mostly comprised of nylon 6, nylon 6.6 from backing made of polypropylene and latex. In the next step, the processing facility takes the baled and sorted waste carpet and processes it to create recycled fiber or plastic materials. Reclaimed nylon can be used in carpet manufacturing while reclaimed plastic materials can be used in other manufacturing processes such as auto parts and construction materials, and recycled-content products such as carpet padding can be directly sold to consumers.

The practical applications of primary or secondary technology require various levels of up-front capital investment, technical knowledge and skills, a labor force, operating costs, and supplies of waste carpet. The primary approach of a pure closed-loop system may require larger capital investment and scale economies, and it may serve multi-state regions in terms of waste carpet collection. In the secondary approach, a small-size facility employing dozens of workers can operate in an economically feasible manner to manufacture carpet underlay, pellet, and molded plastics (Subbiah, 2008). Such a facility can effectively serve a metropolitan economy.

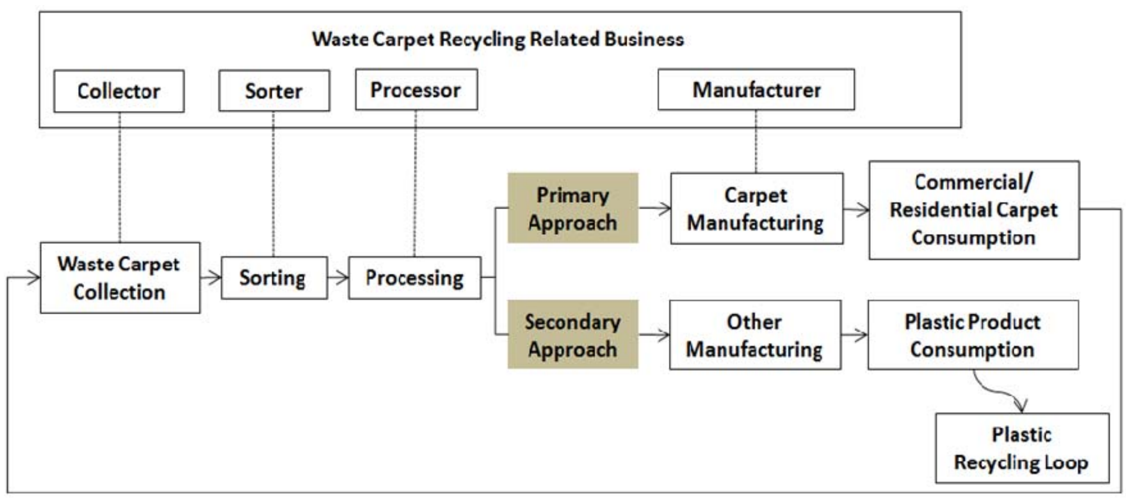

Figure 5: Material Flow of Waste Carpet Recycling

3.3.3. Voluntary Agreement for Waste Carpet Stewardship

The initiatives pertaining to waste carpet recycling by a private company are driven by a voluntary agreement. Historically, product stewardship for waste carpet was initiated by a state environmental agency, the Minnesota Office of Environmental Agency. Then, several mid-western state environmental agencies and the U.S Environmental Protection Agency formed a partnership, the Midwestern Workgroup on Carpet Recycling, for promulgating stewardship of waste carpet (Fishbein, 2000), which the industry interpreted as a warning that it needed to become proactive in carpet recycling. The major carpet manufacturers and the industry association joined the Workgroup and engaged in discussions about the establishment of a carpet recycling system and mechanisms that secure the industry's commitment to diverting and recycling waste carpet. After a two-year negotiation process, representatives of the industry, federal and state governmental agencies, and non-governmental organizations reached an agreement and signed the Memorandum of Understanding for Carpet Stewardship in 2002.

The main focus of the Memorandum of Understanding for Carpet Stewardship was the establishment of a voluntary effort by carpet manufacturers to take physical and financial responsibility while minimizing a role of government in carpet stewardship. According to the meeting notes of the Midwestern Workgroup on Carpet Recycling, the underlying issue was the funding mechanism that potentially determined the role of each participant and the shape of the entire recycling system. The funding mechanism consisted of two options: imposition of a recycling fee on consumers at the point of disposal or sale or the internalization of costs by manufacturers. The first financing option was a more enforceable mechanism requiring the enactment of regulations as well as governmental administrative involvement. The second option was a highly flexible option for industry highlighting the self-regulatory role of OPMs. The workgroup took the idea of a manufacturer responsibility model and excluded direct regulation such as the landfill ban and the disposal fee.[15] As a result, the industry was granted autonomy to flexibly establish its own recycling system.

Industry and government negotiated specific goals and a timeline of the phase-out of disposing waste carpet in landfills. To implement these actions, industry and government agencies agreed to create a third-party organization, the Carpet America Recovery Effort (CARE), funded by the industry. The task of this organization was to strengthen the collection system, to serve as an information source for technology and market development, and to measure and report quantitative progress. In 2002, their efforts led to a negotiated outcome on carpet stewardship in which the diversion goal for the first phase from 2002 to 2012 was established. The diversion rate goal was to be 10% by 2005 and 23% by 2010.

This voluntary agreement scheme for waste carpet recycling among industry, local and federal governmental agencies, and non-governmental organizations is an example of a transition in environmental regulation from a command and control approach to a participatory and consensual approach. The progress of recycling relies on the ability and the willingness of an individual carpet manufacturer and a current market system in which each firm is expected to compete to provide innovative recycling solutions and green products. This voluntary scheme has formed a foundation on which diverse recycling systems in terms of organizational form, geographic scope, and location patterns could be developed by industry. However, the exclusion of mandatory provisions and enforceable mechanisms has been a source of concern because of the lack of progress on the part of manufacturers in their recycling efforts. Indeed, progress in

[15] Source: Minnesota Pollution Control Agency (2000) accessed in October, 2007

waste carpet recycling has been far below negotiated target rates (Carpet America Recovery Effort, 2010).

3.3.4. Industrial Organization and Spatial Linkage for Waste Carpet Recycling

Given the evolution occurring within institutional environments and recycling technology, carpet manufacturers have responded by changing their production systems and industrial organizations. Voluntary agreements regarding waste carpet recycling have allowed individual participants to self-control recycling activities. Because voluntary agreements do not regulate a specific operational and organizational form, diverse private firms, including carpet manufacturers, fiber manufacturers, plastic product manufacturers, and independent small-size carpet recyclers, have formulated a competitive market-based network of recycling systems. Variations in the organizational form and spatial patterning of recycling businesses are analyzed in light of a theoretical model suggested in the previous section. This research categorizes carpet recycling businesses into three groups—in-house recycling, recycling and supply, and small business—by industry segment, organizational type, firm size, applied technology, and spatial pattern, displayed in Table 6.

Table 6: Categories of Business Models for the Carpet Recycling Industry

	In-house Recycling	Recycling & Supply	Small Business
Industry Segment	Carpet manufacturing	Fiber manufacturing; plastic manufacturing	Carpet sorting or processing
Organizational Type	Vertical integration	Specialization	Small business
Firm Size	Large	Large or medium	Small
Applied Technology	*Primary approach:* reclaiming nylon or other fiber	*Primary approach:* reclaiming fiber *Secondary approach:* plastic manufacturing	*Pre-processing:* sorting or physical separation *Secondary approach:* manufacturing plastic products
End-use of Recovered Materials	Used in carpet product: face fiber and backing	Supply to carpet manufacturing or plastic product manufacturing	*Pre-processing:* supply to downstream carpet recycling facility *Secondary approach:* plastics manufacturing or consumer
Spatial Pattern of Recycling Facility	Located near carpet manufacturing facilities in a loosely spatially-confined manner	Proximate to original production line and wider scale of spatial linkage in downstream	Oriented to urban areas

Vertically-integrated carpet manufacturers are a key entity in the carpet recycling market. These manufacturers tended to invest in-house recycling facility and reclaimed recyclable materials in waste carpet such as nylon 6 and nylon 6.6. Reclaimed fibers are internally consumed; that is, vertically-integrated carpet manufacturers produce recycled-content carpet products. They have diversified their product portfolio by incorporating recycling processing into their vertical integration. Since recycling is a part of the integrated carpet manufacturing system, a recycling facility will more likely locate near fiber and carpet manufacturing facilities. However, spatial proximity may not be a top priority because of standardized operations and economies of scale. The location of a recycling facility and other existing manufacturing facilities is loosely confined to a regional scale. As carpet manufacturers cluster in northern Georgia, in-house carpet recycling facilities are more likely to locate in northern Georgia.

The second category is a recycling and supply business model. Because waste carpet contains recyclable materials that can be used in diverse fiber and plastic manufacturing processes, both fiber and plastic manufacturing companies have become involved in carpet recycling. These types of firms reclaim recyclable components and sell their recovered materials or recycled-content products to downstream manufacturers. Currently, reclaimed plastic materials are sold for auto parts and building materials. The spatial implications of this segment are similar to those of in-house recycling, but the spatial linkage of the downstream supply chain is expected to be wider.

The final category is the small business model. Small recycling firms, which operate independently or subcontract with larger carpet recyclers, can manufacture recycled-content products such as carpet underlay, or they can function as a supplier to carpet and fiber manufacturers, for they provide local collecting and sorting services on behalf of carpet manufacturers. Small recycling facilities serve smaller areas such as metropolitan areas or states and have an urban-oriented location pattern for economizing collection costs.

This research presents three illustrative cases of industrial organizations and spatial linkages of waste carpet recycling corresponding to the categories in Table 6. First, Shaw Industries represents a case of in-house recycling and a closed-loop system. Shaw is one of the largest carpet manufacturers in the U.S. It has established a vertically-integrated production system from nylon fiber production, tufted carpet manufacturing, and distribution centers to the waste carpet reclamation. The major carpet manufacturing facilities of Shaw are located in northern Georgia. This case illustrates how waste carpet recycling became a part of a vertically-integrated carpet manufacturer. The primary point at which recycling and carpet manufacturing is linked in Shaw is the Evergreen Nylon Recycling facility located in August, Georgia. Evergreen Nylon Recycling facility was originally constructed as a joint venture of Honeywell and DSM Chemicals in 1999. Because of issues with inefficiency, the facility shut down in 2001. Shaw acquired the Evergreen Nylon Recycling facility in 2005 as part of deal entailing the purchase of nylon fiber manufacturing operations located in South Carolina from Honeywell. After rehabilitation, the facility re-opened in 2007.[16] The operation of the Evergreen recycling facility expanded the processing capacity of waste carpet reclamation; Whereas the total amount of waste carpet recycled at Shaw was 21.1 million pounds in 2006, the amount soared to 94.9 million pounds in 2007 (Shaw Industries Group, Inc. Corporate Sustainability Report, 2009). In 2010, the Evergreen Nylon Recycling facility processed 85 million pounds of waste carpet, bringing the total to 122 million pounds (Carpet America Recovery Effort, 2010).

[16] The history of Evergreen Nylon Recycling facility is drawing from the presentation of David Harless in the 2007 Carpet America Recovery Effort Conference, accessed on July 6, 2011 at http://www.carpetrecovery.org/pdf/annual_conference/2007_conference_pdfs/presentations/Thursday/Harless_ENR.pdf

Altogether, the Shaw recycling facility has hired 70 employees and sales reached $22 million in 2010.[17]

For its integrated recycling system, Shaw has established dozens of its own regional collection sites nationwide to ensure a supply of waste carpet to its recycling facility. At the collection sites, waste carpet is sorted by fiber types and baled carpet shipped to their recycling facility. The primary feedstock used in the Evergreen Nylon Recycling facility is nylon 6 (over 98%).[18] Baled carpet is shredded, grinded into small pieces, and then melted. The vaporized materials are separated into nylon and non-nylon materials. The vapor with nylon material is converted back to liquid form and then into recovered caprolactam in the purification process.[19] The purified caprolactam is shipped to another location of a branch facility that manufactures recycled-content nylon 6, and, eventually the recycled-content carpet tile backing and broadloom carpet under the Shaw brand are manufactured in Georgia. In addition, residuals of other types of materials are used as fuel in the waste-to-energy facility located in Dalton or as other plastic materials further processed by outsourcing firms. The highly standardized production process allows economies of scale in the manufacturing and recycling operations. The spatial links are not confined to a narrow geographical area; instead, facilities are distributed within Georgia and neighboring states, shown in Figure 6.

[17] Dun & Bradstreet Million Dollar Database accessed on July 12, 2011.
[18] Russ DeLozier gave a presentation, a title of "Re-start of evergreen nylon recycling" in 2006 Carpet America Recovery Effort Conference. The powerpoint slides are retrieved from http://www.carpetrecovery.org/pdf/annual_conference/2006_conference_pdfs/Evergreen_Nylon_Recycling_CARE_2006.pdf
[19] The process of the Evergreen Nylon Recycling facility referred to in a video on the Shaw Floors website, accessed on July 12, 2011 at http://www.shawfloors.com/Environmental/RecyclingDetail

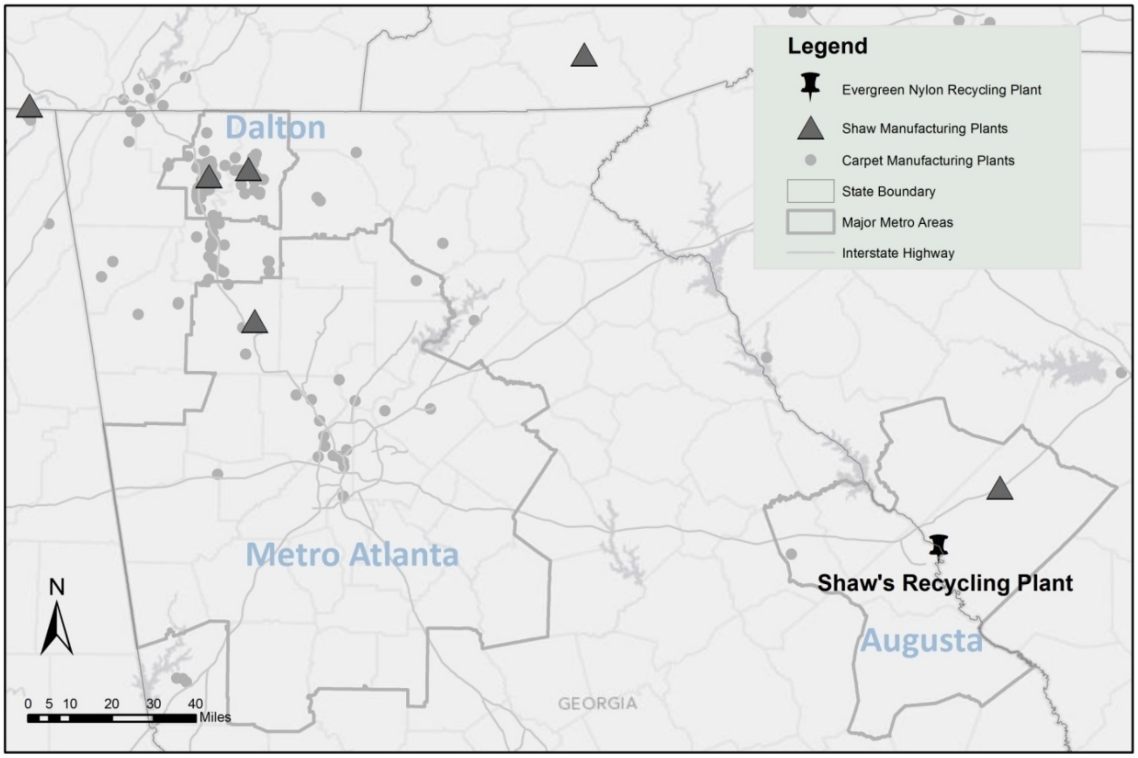

Figure 6: Spatial Distribution of a Closed-loop System in Carpet Recycling by Shaw

Sources: Reference USA and Dun & Bradstreet The Million Dollar Database

The case of Wellman illustrates an open-loop system of waste carpet recycling. Wellman, an expert in the manufacturing of fiber and plastic resin, has also been involved in the recycling of PET bottles since 1970s.[20] Its considerable experience in plastic and fiber recycling has enabled the company to initiate waste carpet recycling since1996. Wellman operates a waste carpet recycling facility located in Johnsonville, South Carolina. This facility converts the nylon material in waste carpet into plastic products. It manufactures engineering resin products such as automotive covers, cooling fans, and air cleaners. In 2005, Wellman shipped 150 million pounds of engineering

[20] The history of Wellman is drawn from the website, Reference for Business, accessed July 13, 2011 at http://www.referenceforbusiness.com/history2/57/Wellman-Inc.html

resin with 25% waste carpet content.[21] The company claims that the recycled-content engineering resin has a cost advantage over virgin resin due to the fluctuating prices of petroleum. The auto manufacturing industry purchases Wellman's recycled-content engine covers. Waste carpet is supplied to Wellman by local collectors, Southeastern Plastics Recovery, a South Carolina-based recycler, and Georgia Carpet Processing. These firms perform waste carpet collecting and pre-processing.

In contrast to large carpet and plastic manufacturing firms, carpet recycling can be a niche market for small business. One example is LA Fiber, located in Vernon, California. In the early 1980s, this company initially recycled pre-consumer textile waste received from the textile industry in the Los Angeles area. After adopting the North America Free Trade Agreement, LA Fiber confronted a problem of securing an adequate supply of pre-consumer textile waste as the textile industry in the Los Angeles area was rapidly declining because of international migration. When recognizing the similarity between textile and carpet recycling and the sufficient feedstock that went into landfills, this company decided to switch to a carpet recycler. With a loan from the Recycling Market Development Zone program run by the California Department of Resources Recycling Recovery, LA Fiber modified its process equipment[22]and began to manufacture carpet padding products from the waste carpet materials. In 2008, the company hired 80 employees and processed 89 million pounds of waste carpet.

---

[21] The Carpet America Recovery Effort website, accessed July 10, 2011 at http://www.carpetrecovery.org/050511_CARE_Recycler_Year.php

[22] California Department of Resources Recycling and Recovery, accessed July 9, 2011 at http://www.calrecycle.ca.gov/Calmax/Inserts/2004/Summer/LAFiber.htm

3.4. Conclusion

The purpose of this chapter is to develop a theoretical model that contributes to the body of knowledge pertaining to the logic of the internal industry organization and the spatial pattern of recycling systems in different institutional and industrial contexts. The theoretical model specifically focuses on the behavior of a firm with regard to industrial recycling activities. The model presented here conceptualizes possible organizational decisions pertaining to recycling and associated location patterns of a recycling facility. This conceptualization is built upon existing theories. In particular, industry organizational theory has provided fundamental insights into the decision making of a responsible company about recycling. Hence, the contribution of this theoretical work is its suggestion of a conceptual framework specified for the organizational and spatial aspects of industrial recycling activities.

The model examined two different systems on the basis of the responsibilities of both local governments and manufacturers. With regard to responsibility of local governments, it indicated that the local ordinance and policy support of these governments as well as the economic logic of vertically-integrated waste management companies played major roles in the development of a localized recycling system. With regard to responsibility of manufacturers, a recycling system develops through a course of reciprocal interaction between industry and regulatory agencies, and various operational rules can be institutionalized. The model suggested possible organizational forms that a responsible manufacturer can take and associated location patterns. Given the considerable autonomous role of industry in manufacturer responsibility, diverse geographic scales of recycling systems can emerge.

The case of CDW recycling in the San Francisco metropolitan area exemplified a localized CDW recycling system with established local ordinances and investment by franchise waste management companies supported by policy incentives. The case of waste carpet recycling illustrated diversified recycling systems built upon the voluntary agreement of waste carpet stewardship in terms of the organizational form and spatial linkage. In-house recycling operated by major carpet manufacturers tended to locate near existing carpet manufacturing facilities, serving a regional or national market. The outsourced or independent small recyclers operated on a smaller geographical scale such as a metropolitan area or a state. The theoretical model and cases revealed the organizational and spatial logic of recycling systems that may be manifested on a local or regional scale.

CHAPTER 4. ANALYTICAL MODEL: ENVIRONMENTAL INPUT-OUPUT MODEL

The purpose of this chapter is to present a regional environmental IO model that estimates the economic and environmental impact of recycling industrial activities. Because various waste management activities such as collection, incineration, recycling, and disposal are typically aggregated in the single sector in the conventional IO model, the addition of a specific recycling industry sector is required in the existing IO framework. The first section of this chapter formally presents a regional environmental IO model that explicitly incorporates a recycling industry sector with a discussion of the features of recycling industry technology that should be considered in IO modeling.

The second section of the chapter addresses the issue of environmental responsibility of a regional economy within a framework of a regional environmental IO model. The open regional economy relies considerably on imported products manufactured outside a particular region to fulfill a function of the regional economy, and conversely, a region may export a number of products to meet demand outside of the region. For attributing environmental burden to a regional economy, the geographic origin of demand and supply should be considered. This section illustrates a typology of the environmental responsibility of a regional economy and shows single- and two-region environmental IO modeling frameworks in accordance with a typology of the environmental responsibility of a regional economy.

4.1. Input-Output Model for the Recycling Industry

The IO model for the recycling industry was developed previously (Choi et al., 2011; Leigh et al., 2012). This section summarizes a key feature of the IO model for the recycling industry presented in the work of Leigh et al. (2012). The commodity-by-industry framework has an inherent advantage in modeling of the recycling industry. Recycling is typically a multi-product process (Suh et al. 2010). For example, most CDW is sent to processing facilities without source separation. A mixed CDW recycling facility processes different types of waste such as concrete, wood, gypsum, and paperboard through mechanical and labor processes with loaders, screens, conveyers, crushers, and grinders. Several recovered products are manufactured through combined processes in a single facility. In other words, recycling is often a non-separable process of multiple outputs.

An industry-by-industry framework does not effectively reflect this inherent feature of the recycling industry because the relationship between industry and commodity in the industry-by-industry framework is a one-to-one match. A secondary product is allocated to an industry that produces a secondary product as a primary product in the industry-by-industry framework (Miller and Blair, 2009). For example, when recycled aggregate manufactured from a recycling process is allocated to the construction sand and gravel mining sector, information about recycled aggregate from processing waste material is combined with that of virgin aggregate. In this case, the flow of recycled materials is difficult to track. Thus, the commodity-by-industry framework, which is capable of separately recording both primary and secondary products, is a superior platform for modeling the recycling industry.

The recycling industry sector and its commodities are specified in the commodity-by-industry framework, shown in Table 7. Because of the nature of the multi-product process of the recycling industry, we need to determine what a primary product is and what a secondary product is. In general, a recycling process is associated with two types of commodities: 1) an intangible service commodity, that is, a waste removal and waste reduction service, and 2) a tangible recovered commodity, which is physically re-consumed in the other production process, or the final demand. In this research, a service commodity is defined as a primary commodity while a recovered commodity is defined as a secondary commodity. The advantage of this arrangement is that it can well reflect financial and physical transactions between entities that generate waste and recycling in the use table (Leigh et al., 2012). The notation is summarized in Table 8.

Table 7: Augmented Commodity-by-Industry Framework for the Recycling Industry

		Industry		Commodity		Final demand	Total output
		j	γ	j	ω		
Industry	i (Conventional)			v_{ij}	$v_{i\omega}$		g_i
	γ (Recycling)			$v_{\gamma j}$	$v_{\gamma\omega}$		g_γ
Commodity	i (Conventional)	u_{ij}	$u_{i\gamma}$			e_i	q_i
	ω (Recycling service)	$u_{\omega j}$	$u_{\omega\gamma}$			e_ω	q_ω
Total output		g_i^T	g_γ^T	q_i^T	q_ω^T		

Table 8: Notation of Variables in the IO Model for the Recycling Industry

Industry and Commodity
$Industry = \{1, \cdots, n, n+1\}$ where industry consists of n numbers of conventional industries, $\{1, \cdots, n\}$, and a recycling industry, $\{1+n\}$
$Commodity = \{1, \cdots, m, m+1\}$, where commodity consists of m numbers of conventional commodities, $\{1, \cdots, m\}$, and a recycling service commodity, $\{m+1\}$
γ : recycling industry is denoted subscript γ
ω: recycling service commodity is denoted subscript ω
Energy and Emissions
$\mathrm{E} = [\mathrm{E}_{kj}]$: energy use of type k by industry j
ε_{kj} : energy use type k by industry j per a dollar worth of output of industry j
$\mathrm{X} = [\mathrm{X}_i]$:greenhouse gas emissions by industry i
χ_i : greenhouse gas emissions by industry i per a dollar worth of output of industry i
Region
r and s : a region is denoted superscripts r and s
$t = [t_i^{rs}]$: flow of commodity i from regions r to s
Variables
$U = [u_{ij}]$: intermediate input commodity demanded by industry j
$V = [v_{ij}]$: output of commodity j produced by industry i
$g = [g_i]$: total output of industry i
$q = [q_i]$: total output of commodity i
$e = [e_i]$: final demand for commodity i (consisting of household consumption, government expenditures, investment, and inventory changes)
$w = [w_j]$: value-added of industry j
$m = [m_j]$: import of commodity j
$x = [x_i]$: export of commodity i
Technical Coefficients
$B = [b_{ij}]$: input commodity purchased by industry j per a dollar worth of output of industry j
$D = [d_{ij}]$: output of commodity j produced by industry i per a dollar worth of output of commodity j

4.1.1. Augmenting Use and Make Tables

Use and make tables are augmented by adding the recycling industry and its commodities. Let $n+1$ be the number of industries with n number of conventional industries and one recycling industry (denoted subscript, γ), and $m+1$ number of commodities with m number of conventional commodities and one recycling service commodity (denoted subscript, ω).

In the use dimension, a recycling service commodity is added in a row, and a recycling industry sector is added in a column. Hence, $u_{\omega j}$ represents the cost of a recycling service that a conventional industry j pays for. The total output of a commodity i and the total output of a recycling service commodity ω is the sum of intermediate and final demand, expressed as a matrix notation, shown in Equation 1.

$$\begin{bmatrix} q_I \\ q_\omega \end{bmatrix} = \begin{bmatrix} U_{IJ} & U_{I\gamma} \\ U_{\omega J} & U_{\omega\gamma} \end{bmatrix} + \begin{bmatrix} e_I \\ e_\omega \end{bmatrix} \tag{1}$$

The use table is transformed into technical coefficients—input commodity i purchased by industry j divided by total industry output g_j ($B_{IJ} = U_{IJ}\hat{g}_J{}^T$), expressed in Equation 2. ^ denotes a diagonal matrix of a vector, and T denotes transposition of the matrix.

$$\begin{bmatrix} q_I \\ q_\omega \end{bmatrix} = \begin{bmatrix} B_{IJ} & B_{I\gamma} \\ B_{\omega J} & B_{\omega\gamma} \end{bmatrix} \begin{bmatrix} g_J \\ g_\gamma \end{bmatrix} + \begin{bmatrix} e_I \\ e_\omega \end{bmatrix} \tag{2}$$

In the make dimension, a recycling service commodity is added in a column, and a recycling industry sector is added in a row. $v_{\gamma j}$ represents the economic value of recovered commodities produced by a recycling industry while $v_{\gamma\omega}$ represents the economic value of a recycling service commodity. The industry output is the sum of all the commodities in a row ($g_I = V_{I\bullet}i$) expressed in Equation 3. i denotes a summation vector in which all elements are 1.

$$\begin{bmatrix} g_I \\ g_\gamma \end{bmatrix} = \begin{bmatrix} V_{IJ} & V_{I\omega} \\ V_{\gamma J} & V_{\gamma\omega} \end{bmatrix} \begin{bmatrix} i \\ i \end{bmatrix} \tag{3}$$

The make table is transformed as the commodity output proportion in which each column element of the make table is divided by commodity total output ($D_{IJ} = V_{IJ}\hat{q}_J{}^T$). The industry output is connected to the commodity output in Equation 4.

$$\begin{bmatrix} g_I \\ g_\gamma \end{bmatrix} = \begin{bmatrix} D_{IJ} & D_{I\omega} \\ D_{\gamma J} & D_{\gamma\omega} \end{bmatrix} \begin{bmatrix} q_J \\ q_\omega \end{bmatrix} \tag{4}$$

4.1.2. Deriving Total Requirement Matrices

Deriving total requirement matrices requires the selection of an appropriate technology assumption between industry-based technology and commodity-based technology assumptions. The industry-based technology assumption indicates that industry has the same fixed input structure for both primary and secondary commodities, indicating that a secondary product is considered a by-product. Alternatively, the commodity-based technology assumption implies that the input structure of a commodity is the same regardless of the production sectors (Miller and Blair, 2009; Jackson et al., 2008; Suh et al., 2010).

As discussed above, recycling is typically a non-separable process over multiple outputs, and the input structure of recycled material differs from that of virgin material. For example, the extent of energy and labor inputs of mining processing for construction gravel and sand and that of a mixed CDW processing facility differ. Hence, the industry-based technology assumption is more appropriate for the IO modeling of the recycling industry.

In order to derive the total requirement matrix, Equation 4 substitutes into the total industry output components in Equation 2, and then it results in Equation 5, in which the commodity demand and the commodity output are linked. The commodity-by-commodity total requirement matrix is expressed in Equation 6, which allows us to compute commodity output changes generated from changes in the commodity demand. Equation 7 shows the industry-by-commodity total requirement matrix that connects commodity demand and industry output. Because most environmental statistics for IO modeling are compiled by industry sectors, the industry-by-commodity total requirement matrix in Equation 7 will be used in the economic and environmental analyses in the following sections.

$$\begin{bmatrix} q_I \\ q_\omega \end{bmatrix} = \begin{bmatrix} B_{IJ} & B_{I\gamma} \\ B_{\omega J} & B_{\omega\gamma} \end{bmatrix} \begin{bmatrix} D_{IJ} & D_{I\omega} \\ D_{\gamma J} & D_{\gamma\omega} \end{bmatrix} \begin{bmatrix} q_J \\ q_\omega \end{bmatrix} + \begin{bmatrix} e_I \\ e_\omega \end{bmatrix} \tag{5}$$

$$\begin{bmatrix} q_I \\ q_\omega \end{bmatrix} = \left(I - \begin{bmatrix} B_{IJ} & B_{I\gamma} \\ B_{\omega J} & B_{\omega\gamma} \end{bmatrix} \begin{bmatrix} D_{IJ} & D_{I\omega} \\ D_{\gamma J} & D_{\gamma\omega} \end{bmatrix} \right)^{-1} \begin{bmatrix} e_I \\ e_\omega \end{bmatrix} \tag{6}$$

$$\begin{bmatrix} g_I \\ g_\gamma \end{bmatrix} = \begin{bmatrix} D_{IJ} & D_{I\omega} \\ D_{\gamma J} & D_{\gamma\omega} \end{bmatrix} \left(I - \begin{bmatrix} B_{IJ} & B_{I\gamma} \\ B_{\omega J} & B_{\omega\gamma} \end{bmatrix} \begin{bmatrix} D_{IJ} & D_{I\omega} \\ D_{\gamma J} & D_{\gamma\omega} \end{bmatrix} \right)^{-1} \begin{bmatrix} e_I \\ e_\omega \end{bmatrix} \tag{7}$$

4.2. Regional Environmental Input-Output Model

The regional environmental IO model serves as an analytical tool that examines the direct and indirect regional environmental impact in terms of energy use and GHG emissions. The environmental impact of one region is spatially linked to other regions through inter-regional trade. This section proposes a typology of environmental responsibility of a regional economy associated with the geographic origin of demand and supply, and then it presents four environmental modeling frameworks matching the concept of the environmental responsibility of a regional economy in the single- and two-region approaches.

4.2.1. Environmental Responsibility of a Regional Economy

In an open regional economy, a large number of products and materials are imported and exported to meet the demand of both regional industries and consumers. The inter-regional trade of products and services contributes to a substantial amount of environmental emissions. The burden that regional industrial activities place on the environment is often driven by the demands of other regions. Hence, the environmental responsibility of a region differs according to the accepted principle of whether it is demand or supply oriented. Therefore, the clarification of environmental responsibility related to inter-regional flows of products is an important task in regional environmental IO modeling. The box (A) in Figure 7 shows three types of emissions and the role of export and import in the attribution of environmental responsibility to a regional economy.

The origins of demand and supply characterize the types of emissions. Type 1 consists of emissions generated from regional production activities driven by regional demand. For example, to meet the demand of local residents, local food processing facilities directly emit GHGs from the combustion of natural gas. Second, to meet the demand from outside a region, regional industry activities might generate an environmental burden; that is Type 2 emissions which are generated from regional production activities driven by out-of-region demand. It is an export-related environmental burden. Finally, Type 3 emissions are generated from out-of-region production activities driven by regional demand; that is, they are emissions related to imported products in which actual emissions are released outside of a region. The combination of these three emissions types yields four concepts of regional environmental responsibility: territorial environmental responsibility, regional consumption-based environmental responsibility, regional production-based environmental responsibility, and full regional environmental responsibility.

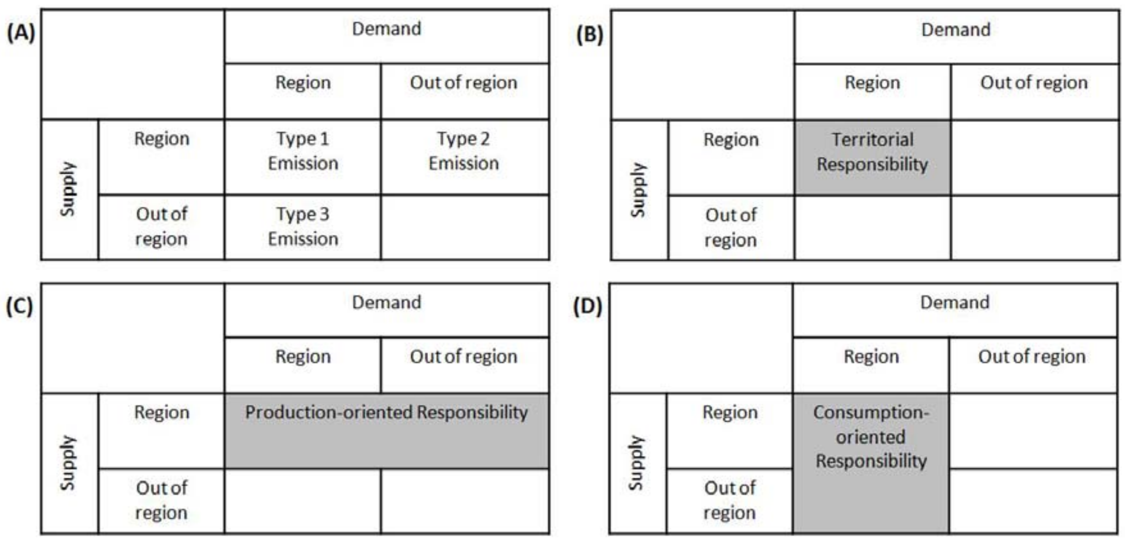

Figure 7: Typology of Regional Environmental Responsibility

1) ***Territorial environmental responsibility***: A region is responsible only for environmental burden that it can directly control in terms of consumption and production. This concept does not include the environmental burden originating from any out-of-region industrial and consumption activities. Territorial environmental responsibility includes only Type 1 emissions.

2) ***Regional consumption-based environmental responsibility***: In this concept, consumption is the ultimate cause of all production activities and associated emissions. Therefore, a region is responsible for the environmental burden originating from all regional consumption demands, regardless of the origin of production. Whereas environmental burden pertaining to regional production activities for exports are excluded in this category, environmental burden originating from imported products are included. Thus, Type 1 and 3 emissions are a regional consumption-based environmental responsibility.

3) *Regional production-based environmental responsibility*: A region is responsible for the environmental burden directly generated by production activities within a region. Only environmental burden occurring within a region is attributed to regional responsibility. Environmental burden pertaining to imported products is excluded in this approach. Thus, Type 1 and 2 emissions are a regional production-based environmental responsibility.

4) *Full regional environmental responsibility*: A region is responsible for any emissions pertaining to any regional production and consumption activity. Therefore, it includes Type 1, 2, and 3 emissions.

This typology clarifies the relationship between the cause of environmental burden and actual emissions, and which environmental burdens can be attributed to a regional economy.

4.2.2. Single-Region Environmental Input-Output Modeling Approach

This section lays out the table of a single-region environmental IO model and presents single-region environmental IO modeling frameworks with respect to the four principles of environmental responsibility of a regional economy.

Regional Environmental IO Model with Regional Technical Coefficients

A single region is denoted by superscript, r. Let $g^r = [g_j^r]$ denote the regional output of industry j and $e^r = [e_i^r]$ the regional final demand of commodity i. The final demand includes household consumption, government expenditures, investment, and inventory changes. The regional use table is denoted $U^r = [u_{ij}^r]$, which represents intermediate input commodity i demanded by industry j. This regional use table does not distinguish the origin of commodity supply regions. The use table is re-stated as regional technical coefficients, input commodity i of industry j divided by regional output of industry j, where $B^r = [b_{ij}^r]$ and $B_{IJ}^r = U_{IJ}^r \hat{g}_J^{rT}$. As an identity equation, the regional total commodity output is equal to regional intermediate and final demand and export.

$$q^r = U^r + e^r + x^r = B^r g^r + e^r + x^r \tag{8}$$

The regional make table is denoted $V^r = [v_{ij}^r]$, which represents commodity output j produced by industry i. The make table is re-stated by either the commodity output proportion or the industry output proportion. The form of the total requirement matrix in the commodity-by-industry framework will differ according to the choice between the assumptions of commodity technology and industry technology. As noted above, because this research focuses on the recycling industry, the industry technology assumption is taken. The commodity output proportions are calculated by each element of the make table divided by total commodity output j where $D^r = [d_{ij}^r]$ and $D_{IJ}^r = V_{IJ}^r \hat{q}_J^{rT}$. The column sum of the make table represents the commodity supply within a region, and the row sum of the make table shows the total regional industry production.

$$V^r i = q^r \text{ and } i^T V^r = g^{rT} \tag{9}$$

In the U.S benchmark input-output model, an import is a negative entry of the final demand. In the regional model, an import is transposed and separately added as a new row, and it is emphasized in the regional input-output model because a regional economy is more open than a national economy. Trade among regions is an important parameter that determines regional economic and environmental impact (Jackson, 1998). Import is denoted $m^r = [m_i^r]$, which represents import commodity i in region r. As an identity equation, the total regional commodity supply is the sum of a regionally-produced commodity and an imported commodity.

$$s^{rT} = i^T V^r + m^T \tag{10}$$

The physical measures of resource use and GHG emissions are added in the conventional regional commodity-by-industry framework. Energy use is denoted $\mathrm{E}^r = [\mathrm{E}_{kj}^r]$, which represents regional energy use of type k in industry j. As energy is one of the intermediate input commodities in industry production, it is added in the row. The energy use coefficient is defined as energy use of type k by industry j per a dollar worth of output of industry j, $\varepsilon_{kj}^r = \mathrm{E}_{kj}^r \hat{g}_j^{rT}$ .GHG emissions are denoted $\mathrm{X}^r = [\mathrm{X}_i^r]$, which represents regional GHG emissions of industry i. Environmental emissions are added in the column, for they are treated as an undesirable by-product of industry production emitted into the environment. The GHG emission coefficient is defined as the GHG emissions of industry i per a dollar worth of output of industry i, $\chi_i^r = \mathrm{X}_i^r \hat{g}_i^{rT}$.

Table 9: Single-Region Environmental Input-Output Table with Regional Technical Coefficients

	Industry	Commodity	Final Demand	Export	Total Output	GHG Emissions
Industry		V^r			g^r	X^r
Commodity	U^r		e^r	x^r	q^r	
Value-added	w^r					
Import		m^T				
Total Outlay (Total Supply)	g^{rT}	s^{rT} ($q^{rT} = s^{rT} - m^T$)				
Energy Resource	E^r					

Source: Extended from Jackson (1998)

Regional Environmental IO Model with Intra-Regional Input Coefficients

Regional technical coefficients do not distinguish whether commodity inputs are regionally produced or imported. If imported commodities are removed from the use table, the table will contain regionally supplied intermediate inputs. Let $U^{rr} = [U_{ij}^{rr}]$ denote regionally-supplied intermediate input commodity i used by industry j and $U^{\bullet r} = [U_{ij}^{\bullet r}]$ the imported intermediate input commodity i used by industry j. The imported commodities for the intermediate inputs are separated from the regionally-supplied intermediate input commodities shown in Table 10. The intra-regional input coefficients are defined as regionally supplied input commodity i of industry j divided by the regional output of industry j, where $B^{rr} = [b_{ij}^{rr}]$ and $B_{IJ}^{rr} = U_{IJ}^{rr} \hat{g}_J^{rT}$.

Table 10: Single-Region Environmental Input-Output Table with Intra-Regional Input Coefficients

	Industry	Commodity	Final Demand	Export	Total Output	GHG Emissions
Industry		V^r			g^r	X^r
Commodity	U^{rr}		e^r	x^r	q^r	
Imported Commodity	$U^{\cdot r}$					
Value-added	w^r					
Import		m^T				
Total Outlay (Total Supply)	g^{rT}	s^{rT} $(q^{rT} = s^{rT} - m^T)$				
Energy Resource	E^r					

Source: Extended from Jackson and Schwarm (2011)

Environmental Input-Output Modeling of the Single-Region Approach

Based on the tables of a single-region environmental IO model, this research derives four environmental IO modeling frameworks with respect to the four types of regional environmental responsibility. The total requirement matrices can take several forms in the commodity-by-industry framework. For simplicity, the industry-by-commodity total requirement matrix using the technology-based assumption is used for illustration of this environmental modeling. Four environmental IO modeling frameworks differ with regard to the compositions of the final demand term (i.e., regional consumption of a regionally-produced commodity, regional consumption of an imported commodity, and an export) and the choice of technical coefficients (i.e., regional technical coefficients and intra-regional input coefficients).

1) *Modeling for regional consumption-based environmental responsibility:* This concept of responsibility accounts for all types of environmental burden driven by all types of regional consumption regardless of the origin of production. Regional emissions occurring during the production activities for export are excluded. Hence, the final demand term in consumption-based environmental responsibility includes only the regional consumption of both regionally produced and imported commodities, $e^r = [e_i^r]$, including household consumption, government expenditures, investment, and inventory changes. The commodity final demand is transformed into industry final demand pre-multiplied by the commodity output proportion matrix, $D^r e^r$. Direct energy use and direct emissions by industry in the regional consumption-based environmental responsibility are expressed as

$$\mathrm{E}^r_{Direct} = \varepsilon^r D^r (e^r) \text{ and } \mathrm{X}^r_{Direct} = \chi^r D^r (e^r) \tag{11}$$

To calculate total (direct and indirect) energy use and GHG emissions, we need to choose one of either the regional technical coefficients or the intra-regional input coefficients. Since the regional consumption-based environmental responsibility conceptually encompasses energy use and emissions related to intermediate inputs produced out of region but consumed within a region, the regional technical coefficients are used. The total consumption-based regional environmental impact is expressed as

$$\mathrm{E}^r_{Total} = \varepsilon^r D^r (I - B^r D^r)^{-1} e^r \text{ and } \mathrm{X}^r_{Total} = \chi^r D^r (I - B^r D^r)^{-1} e^r \tag{12}$$

2) *Modeling for regional production-based environmental responsibility:* This concept of responsibility includes environmental burden from regional production activities that meet regional demand and export. It excludes environmental burden generated from the production of imported commodities for regional intermediate and final demand. Regional consumption is decomposed into regionally-produced commodities and imported commodities, denoted e^{rr} and $e^{\bullet r}$, respectively. The regional commodity export is denoted $x^r = [x_i^r]$. The final demand term of the production-based approach is the sum of the regional consumption demand of regionally-produced commodity and export $e^{rr} + x^r$. Direct energy use and direct GHG emissions in regional production-based environmental responsibility are expressed as

$$\mathrm{E}^r_{Direct} = \varepsilon^r D^r (e^{rr} + x^r) \text{ and } \mathrm{X}^r_{Direct} = \chi^r D^r (e^{rr} + x^r) \quad (13)$$

The intra-regional input coefficients are used for calculating the total emissions of regional production-based environmental responsibility because it excludes emissions generated from the production of intermediate input commodities outside a region. The total regional production-based environmental impact is expressed as

$$\mathrm{E}^r_{Total} = \varepsilon^r D^r (I - B^{rr} D^r)^{-1} (e^{rr} + x^r) \text{ and } \mathrm{X}^r_{Total} = \chi^r D^r (I - B^{rr} D^r)^{-1} (e^{rr} + x^r) \quad (14)$$

3) *Modeling for territorial environmental responsibility:* This concept of responsibility represents a subset of consumption- and production-based approaches. It includes only environmental burden exerted from regionally-produced commodities

consumed inside a region. The final demand term is defined as the consumption of regionally-produced commodity e^{rr}. Direct energy use and GHG emissions by industry in territorial environmental responsibility is expressed as

$$\mathrm{E}^r_{Direct} = \varepsilon^r D^r (e^{rr}) \text{ and } \mathrm{X}^r_{Direct} = \chi^r D^r (e^{rr}) \tag{15}$$

The intra-regional technical coefficients are used for calculating the total emissions of the territorial environmental responsibility because it excludes emissions resulting from out-of-region production. The territorial environmental impact is expressed as

$$\mathrm{E}^r_{Total} = \varepsilon^r D^r (I - B^{rr} D^r)^{-1} (e^{rr}) \text{ and } \mathrm{X}^r_{Total} = \chi^r D^r (I - B^{rr} D^r)^{-1} (e^{rr}) \tag{16}$$

4) *Modeling for full regional environmental responsibility*: This concept of responsibility encompasses all types of environmental burden pertaining to regional consumption and production activities regardless of the geographic origin of production. The final demand term includes all regional consumption of both regionally-produced and imported commodities, and exports. The final demand term is $e^r + x^r$. Direct energy use and direct emissions in the full regional environmental responsibility are expressed as

$$\mathrm{E}^r_{Direct} = \varepsilon^r D^r (e^r + x^r) \text{ and } \mathrm{X}^r_{Direct} = \chi^r D^r (e^r + x^r) \tag{17}$$

The regional technical coefficients are used to account for all environmental burdens. The full regional environmental impact is expressed as

$$\mathrm{E}^{r}_{Total} = \varepsilon^{r} D^{r} (I - B^{r} D^{r})^{-1} (e^{r} + x^{r}) \text{ and } \mathrm{X}^{r}_{Total} = \chi^{r} D^{r} (I - B^{r} D^{r})^{-1} (e^{r} + x^{r}) \quad (18)$$

Each environmental modeling framework is summarized in Table 11. The drawback of a single-region approach is that it does not consider variations in the input structure and energy use patterns of different regions, particularly in consumption-based and full regional environmental responsibility. The two-region IO modeling approach illustrated in the next section provides a more accurate framework.

Table 11: Summary of Regional Environmental Responsibility and Environmental Impact Modeling Based on the Single-Region Input-Output Model

		Territorial Environmental Responsibility	**Regional Consumption-Based Environmental Responsibility**
Technical Coefficients		Intra-regional input coefficients	Regional technical coefficients
Final Demand Term		Regionally-produced and consumed commodities	Regionally-produced and imported commodities
Direct	Energy Use	$E^r_{Direct} = \varepsilon^r D^r (e^{rr})$	$E^r_{Direct} = \varepsilon^r D^r (e^r)$
	Emissions	$X^r_{Direct} = \chi^r D^r (e^{rr})$	$X^r_{Direct} = \chi^r D^r (e^r)$
Total	Energy Use	$E^r_{Total} = \varepsilon^r D^r (I - B^{rr} D^r)^{-1} (e^{rr})$	$E^r_{Total} = \varepsilon^r D^r (I - B^r D^r)^{-1} e^r$
	Emissions	$X^r_{Total} = \chi^r D^r (I - B^{rr} D^r)^{-1} (e^{rr})$	$X^r_{Total} = \chi^r D^r (I - B^r D^r)^{-1} e^r$
		Regional Production-Based Environmental Responsibility	**Full Regional Environmental Responsibility**
Technical Coefficients		Intra-regional input coefficients	Regional technical coefficients
Final Demand Term		Regionally produced and consumed commodities and exported commodities	All commodities regionally produced, imported, and exported
Direct	Energy Use	$E^r_{Direct} = \varepsilon^r D^r (e^{rr} + x^r)$	$E^r_{Direct} = \varepsilon^r D^r (e^r + x^r)$
	Emissions	$X^r_{Direct} = \chi^r D^r (e^{rr} + x^r)$	$X^r_{Direct} = \chi^r D^r (e^r + x^r)$
Total	Energy Use	$E^r_{Total} = \varepsilon^r D^r (I - B^{rr} D^r)^{-1} (e^{rr} + x^r)$	$E^r_{Total} = \varepsilon^r D^r (I - B^r D^r)^{-1} (e^r + x^r)$
	Emissions	$X^r_{Total} = \chi^r D^r (I - B^{rr} D^r)^{-1} (e^{rr} + x^r)$	$X^r_{Total} = \chi^r D^r (I - B^r D^r)^{-1} (e^r + x^r)$

4.2.3. Two-Region Environmental Input-Output Modeling Approach

The multi-region environmental IO model can provide an integrative approach suited for the concepts of regional environmental responsibility. In this section, two-region environmental IO modeling frameworks are specified using the multi-regional approach.[23]

[23] The inter-regional approach and multi-regional approach were developed in previous studies. The required data for the interregional approach were difficult to obtain, particularly in terms of industry destination of inter-regional commodity trade. Alternatively, a multi-regional approach was developed (Miller and Blair, 2009), and the IMPLAN utilized the multi-regional approach.

Let us assume two regions, r and s in a nation, and the rest of world. The commodity trade is an important component in the construction of a multi-regional IO model. Let t_i^{rr}, t_i^{sr}, and $t_i^{\bullet r}$ denote the flow of commodity i into region r from region r, from region s, and from the rest of world, respectively. The total flow of commodity i into region r is $T_i^r = T_i^{rr} + T_i^{sr} + T_i^{\bullet r}$. Then, the proportion of commodity flow coming from each region can be obtained through dividing the commodity flow from one region by the total commodity flow $t_i^{sr} = T_i^{sr} / T_i^r$. The two-region case has six pairs of commodity flow tables that show the proportion of the geographic origin of commodity i.

$$t^{rr} = \begin{bmatrix} t_1^{rr} \\ \vdots \\ t_m^{rr} \end{bmatrix},\ t^{sr} = \begin{bmatrix} t_1^{sr} \\ \vdots \\ t_m^{sr} \end{bmatrix},\ t^{\bullet r} = \begin{bmatrix} t_1^{\bullet r} \\ \vdots \\ t_m^{\bullet r} \end{bmatrix},\ t^{ss} = \begin{bmatrix} t_1^{ss} \\ \vdots \\ t_m^{ss} \end{bmatrix},\ t^{rs} = \begin{bmatrix} t_1^{rs} \\ \vdots \\ t_m^{rs} \end{bmatrix},\ \text{and } t^{\bullet s} = \begin{bmatrix} t_1^{\bullet s} \\ \vdots \\ t_m^{\bullet s} \end{bmatrix} \qquad (19)$$

where m is the number of commodities.

The regional use table, regardless of the geographic origin of the commodity supply, is modified by the commodity flow tables for estimating the intra- and inter-regional input coefficients for each region. Let U^r and U^s denote regional use tables regardless of the geographic origin of the commodity supply. The regional use tables are pre-multiplied by the diagonal matrix of each commodity flow table expressed in Equation 20.

$$\begin{bmatrix} \hat{t}^{rr} & \hat{t}^{rs} \\ \hat{t}^{sr} & \hat{t}^{ss} \\ \hat{t}^{\bullet r} & \hat{t}^{\bullet s} \end{bmatrix} \begin{bmatrix} U^r & 0 \\ 0 & U^s \end{bmatrix} = \begin{bmatrix} \hat{t}^{rr}U^r & \hat{t}^{rs}U^s \\ \hat{t}^{sr}U^r & \hat{t}^{ss}U^s \\ \hat{t}^{\bullet r}U^r & \hat{t}^{\bullet s}U^s \end{bmatrix} \tag{20}$$

Each element on the right side of Equation 20 represents the consumption of intermediate input commodities that are intra-regionally traded, inter-regionally imported, and internationally imported to regions r and s. Technically, the regional use table is row-adjusted by the proportion of commodity origins in terms of intra-regional, inter-regional, and international inflow. The intra-regional use tables are denoted $\hat{t}^{rr}U^r = \tilde{U}^{rr}$ and $\hat{t}^{ss}U^s = \tilde{U}^{ss}$, and the inter-regional use tables $\hat{t}^{sr}U^r = \tilde{U}^{sr}$ and $\hat{t}^{rs}U^s = \tilde{U}^{rs}$. The total commodity output in region r is the sum of the intermediate input demand of regions r and s, the final demand of regions r and s, and the export expressed in Equation 21. Each use table is restated as the input commodity proportion, that is, input commodity i of industry j divided by the regional output of industry j where $\tilde{B}^{rr} = [\tilde{b}_{ij}^{rr}]$ and $\tilde{b}_{ij}^{rr} = \tilde{u}_{ij}^{rr}\hat{g}_j^{r^T}$.

$$q^r = \tilde{U}^{rr}i + \tilde{U}^{rs}i + \hat{t}^{rr}e^r + \hat{t}^{rs}e^s + x^r \tag{21}$$

The regional make table is treated in the same manner as it was in the single-regional model. Domestic trade and foreign imports are distinguished in a two-region approach. A domestic import from region s to r is denoted m^{sr}, and foreign import to region r is denoted $m^{\bullet r}$. These imports are recorded in a new row. The total supply of region r is the sum of regional production and domestic and international imports.

$$s^{r} = i^{T}V^{r} + m^{sr} + m^{\bullet r} \tag{22}$$

Finally, energy use and GHG emissions are added in the same manner as they were in a single-region approach. Energy use in region r and s is denoted E^{r} and E^{s} respectively, and GHG emissions in region r and s are denoted X^{r} and X^{s} respectively.

Table 12: Multi-regional Environmental Input-Output Table

		Commodity		Industry		Final Demand	Export	Total Output	GHG Emissions
		Region r	Region s	Region r	Region s				
Commodity	Region r			$\hat{t}^{rr}U^{r}$	$\hat{t}^{rs}U^{s}$	e^{r}	x^{r}	q^{r}	
	Region s			$\hat{t}^{sr}U^{r}$	$\hat{t}^{ss}U^{s}$	e^{s}	x^{s}	q^{s}	
Industry	Region r	V^{r}						g^{r}	X^{r}
	Region s		V^{s}					g^{s}	X^{s}
Value-added				w^{r}	w^{s}				
Domestic Import		m^{srT} $(= x^{sr})$	m^{rsT} $(= x^{rs})$						
Foreign Import		$m^{\bullet rT}$	$m^{\bullet sT}$	$\hat{t}^{\bullet r}U^{r}$	$\hat{t}^{\bullet s}U^{s}$				
Total Outlay (Total Supply)		s^{rT}	s^{sT}	g^{rT}	g^{sT}				
Energy Resource				E^{r}	E^{s}				

Environmental Input-Output Modeling frameworks of the Two-Region Approach

Using a two-region environmental IO table, this research derives four environmental IO modeling frameworks with respect to the four concepts of environmental responsibility of a region. The industry-by-commodity total requirement matrix of two-region environmental IO modeling is expressed in Equation 23. The final demand differentiates the environmental modeling suited to the four concepts.

$$\begin{bmatrix} g^r \\ g^s \end{bmatrix} = \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} & \hat{t}^{rs} \\ \hat{t}^{sr} & \hat{t}^{ss} \end{bmatrix} \begin{bmatrix} e^r \\ e^s \end{bmatrix} \tag{23}$$

Since the final demand term in territorial environmental responsibility includes only the consumption of regionally-produced commodities, it is expressed as $\begin{bmatrix} \hat{t}^{rr} e^r \\ 0 \end{bmatrix}$. The final demand term in regional production-based environmental responsibility is expressed as $\begin{bmatrix} \hat{t}^{rr} e^r + x^r \\ 0 \end{bmatrix}$ because it includes the consumption of regionally produced commodities and exports. Regional consumption-based environmental responsibility includes the consumption of commodities produced both inside and outside of a region, so the final demand term is split into two components, each corresponding to a region, expressed as $\begin{bmatrix} \hat{t}^{rr} e^r \\ \hat{t}^{sr} e^r \end{bmatrix}$. Finally, the final demand term in full environmental responsibility encompasses the regional consumption of a regionally-produced and imported commodity and export, which is expressed as $\begin{bmatrix} \hat{t}^{rr} e^r + x^r \\ \hat{t}^{sr} e^r \end{bmatrix}$. By combining the total requirement matrix and final demand terms, total energy use and GHG emissions in the four principles of regional environmental responsibility are derived, shown in Table 13.

Table 13: Summary of Environmental Input-Output Modeling with a Two-Region Approach

Territorial Environmental Responsibility	
Total Energy Use	$\begin{bmatrix} E^r \\ E^s \end{bmatrix} = \begin{bmatrix} \varepsilon^r \\ \varepsilon^s \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} e^r \\ 0 \end{bmatrix}$
Total GHG Emissions	$\begin{bmatrix} X^r \\ X^s \end{bmatrix} = \begin{bmatrix} \chi^r \\ \chi^s \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} e^r \\ 0 \end{bmatrix}$
Regional Production-Based Environmental Responsibility	
Total Energy Use	$\begin{bmatrix} E^r \\ E^s \end{bmatrix} = \begin{bmatrix} \varepsilon^r \\ \varepsilon^s \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} e^r + x^r \\ 0 \end{bmatrix}$
Total GHG Emissions	$\begin{bmatrix} X^r \\ X^s \end{bmatrix} = \begin{bmatrix} \chi^r \\ \chi^s \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} e^r + x^r \\ 0 \end{bmatrix}$
Regional Consumption-Based Environmental Responsibility	
Total Energy Use	$\begin{bmatrix} E^r \\ E^s \end{bmatrix} = \begin{bmatrix} \varepsilon^r \\ \varepsilon^s \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} e^r \\ \hat{t}^{sr} e^r \end{bmatrix}$
Total GHG Emissions	$\begin{bmatrix} X^r \\ X^s \end{bmatrix} = \begin{bmatrix} \chi^r \\ \chi^s \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} e^r \\ \hat{t}^{sr} e^r \end{bmatrix}$
Full Regional Environmental Responsibility	
Total Energy Use	$\begin{bmatrix} E^r \\ E^s \end{bmatrix} = \begin{bmatrix} \varepsilon^r \\ \varepsilon^s \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} e^r + x^r \\ \hat{t}^{sr} e^r \end{bmatrix}$
Total GHG Emissions	$\begin{bmatrix} X^r \\ X^s \end{bmatrix} = \begin{bmatrix} \chi^r \\ \chi^s \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \left(I - \begin{bmatrix} \tilde{B}^{rr} & \tilde{B}^{rs} \\ \tilde{B}^{sr} & \tilde{B}^{ss} \end{bmatrix} \begin{bmatrix} D^r & 0 \\ 0 & D^s \end{bmatrix} \right)^{-1} \begin{bmatrix} \hat{t}^{rr} e^r + x^r \\ \hat{t}^{sr} e^r \end{bmatrix}$

CHAPTER 5. DATA AND ENVIRONMENTAL INVENTORY

The previous study indicated the usefulness of a sub-national environmental IO model for sustainable development planning (Munday and Roberts, 2006). However, practical development of a regional environmental IO model has been thwarted primarily by lack of region-specific data. Owing to the dearth of region-specific statistics, this research first seeks to estimate the national-level energy use coefficients and GHG emission coefficients. Then, it explores the possibility of the regionalization of energy use and GHG emission coefficients. This chapter explains the sources of data and practical steps toward constructing a national and regional energy use and GHG emissions inventory and discusses the uncertainties and limitations.

5.1. National Environmental Inventory

The first task is to estimate the national-level energy use and GHG emission coefficients. The development of a national-level energy use and GHG emissions inventory was initially pioneered by the Green Design Institute for the years 1992, 1997, and 2002. The various sources of energy use data were explored and compiled by the Green Design Institute. Webber et al. (2009) documented the data sources for the 2002 EIO-LCA model. Using the Green Design Institute documents, this research updates the national-level energy use and GHG emission coefficients to the year 2006, which are utilized for regionalization. Since one of the disadvantages of an IO-based environmental analysis is its dependency on outdated sources (Hendrickson et al., 2006), an update of recent statistics could reduce estimation errors in the calculation of environmental impact. In this regional environmental IO model, the base year is 2006, for which valid data for an update are available. For example, the Manufacturing Energy Consumption Survey (MECS) contains important statistics that inform industrial energy consumption, and the most recent published data of the MECS come from the year 2006.

5.1.1. Energy Use

Although energy use statistics for the U.S. national economy are relatively plentiful, construction of a consistent energy use dataset across industry sectors poses several challenges. The industry classifications of IMPLAN and energy use statistics are not well matched, and their statistics cover different fuel types measured by different units such as dollar value, physical quantity, or energy content. In general, the industry classifications of energy use statistics are not as sufficiently specified as those of IMPLAN, so statistical data compiled from diverse sources must be adjusted to fit the IMPLAN industry classifications. Therefore, the construction of an energy use inventory by industry sector requires a disaggregation of energy use statistics, unit conversion, and allocation. The primary data sources for a national energy use inventory are summarized by sectoral group in Table 14. The remainder of this section describes the data sources and disaggregation and allocation procedures by sectoral group.

Table 14: Data Sources for Energy Use by Sectoral Group

Sectoral Group	Sources	Type of Covered Energy (Units)	IMPLAN Sector Code
Agriculture	Census of Agriculture 2007	Aggregated energy expenditures ($): gasoline, fuel, and oil	1 – 13
Mining	Economic Census 2007 (Sector 21: Mining/ Subject Series: Materials Summary/ Selected Supplies)	Six categories of energy types with physical units: electricity (kWh), coal (Short ton), natural gas (1,000 cu ft), diesel/gasoline/residual fuel (in gallons)	19-29
Utility	Annual Energy Review 2006	Energy consumption for electricity generation: coal/natural gas/petroleum (in trillions of Btu)	30/495/498
Manufacturing	Manufacturing Energy Consumption Survey 2006; Economic Census 2007 (Industry Series Manufacturing: Electricity)	13 Categories of energy: electricity, residual fuel oil, distillate fuel oil, natural gas, coal, coke and breeze, coke oven gas, waste gas, petroleum gas, pulping liquor, wood chips, waste oil and materials (in trillions of Btu); electricity (1,000kWh)	46- 389
Transportation	Transportation Energy Data Book 27 Edition	Transportation energy by mode: gasoline, diesel fuel, LPG, residual fuel oil, natural gas, electricity (in trillions of Btu)	391 – 397
Federal Government	Annual Energy Review 2006	U.S. government energy consumption: coal, natural gas, aviation gas, distillate/residual fuel oil, jet fuel, motor gasoline, LPG, electricity (in trillions of Btu)	398/505/506
Others	IMPLAN 2006 National Use Table	Commodity purchase from energy producing sectors (in millions of dollars) Coal: coal mining sector Electricity: power generation and supply sector Natural gas: natural gas distribution sector Petroleum-based fuel: petroleum refineries sector	14-18 (Forestry) 33 -45 (Construction) 390/399-494 (Service) 495- 509 (Government except federal and local utility-related sector)

Agriculture Sector: Energy expenditure information for 14 agricultural sectors is available in the Census of Agriculture 2007. However, the Census of Agriculture 2007 provides only aggregate energy consumption with dollar value for each agricultural sector. Detailed energy consumption by fuel types such as gasoline, diesel, natural gas, and electricity was available only in the Census of Agriculture 1997, from which the ratio of energy consumption by fuel type is applied to disaggregate the aggregate energy expenditure of 2007. After disaggregation by fuel type, the dollar value of energy consumption was converted into a physical quantity by dividing it by each fuel price. The Agricultural Statistics 2007, published by the U.S. Department of Agriculture, documented the average price of gasoline, diesel, and LPG used in the agricultural sectors. Finally, the physical unit of energy consumption is converted into an energy unit. The energy-content information for individual fuel came from the energy conversion calculator of the U.S. Energy Information Administration (U.S. EIA).[24] The ratios of energy consumption by fuel type in the agricultural sectors are displayed in Figure 8. Petroleum-based fuel is the major energy source of most agricultural sectors. A larger proportion of natural gas is consumed in greenhouse and nursery production.

[24] Energy Conversion Calculator, U.S. Energy Information Administration website http://www.eia.gov/energyexplained/index.cfm?page=about_energy_conversion_calculator#mogascalc

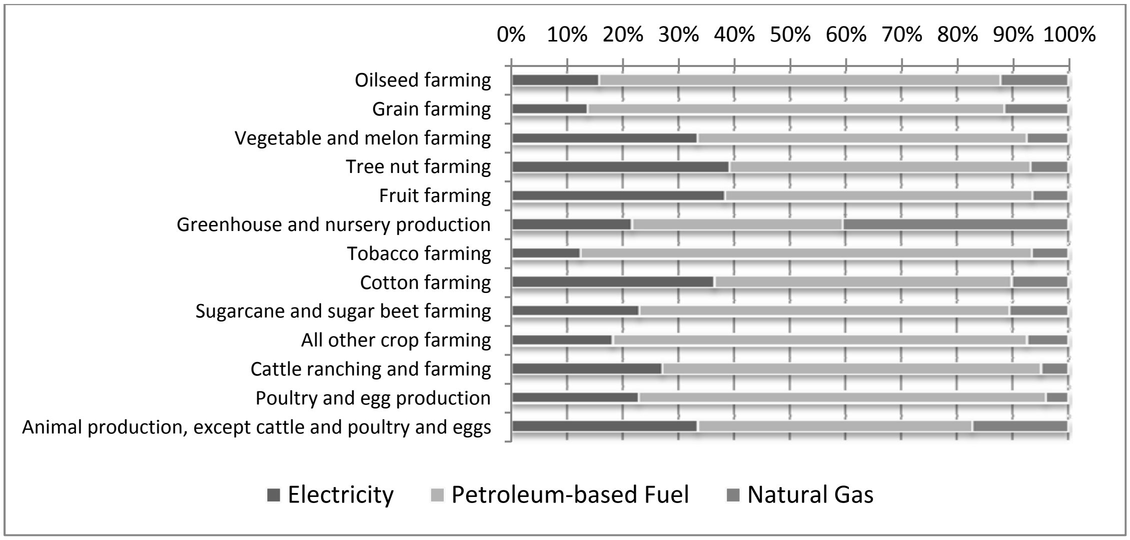

Figure 8: Composition of Energy Use by Fuel Type in the Agricultural Sector

Source: Census of Agriculture 2007

Mining Sector: Economic Census 2007(Mining/Subject Series: Materials Summary) provides information pertaining to energy use with physical units for 11 mining sectors. It includes several fuel types: electricity, coal, natural gas, diesel, gasoline, and residual fuel oil. The physical quantity of fuel consumption is converted into energy units. The energy use data of the 11 mining sectors are recombined into nine mining sectors of IMPLAN. The composition of energy consumption by fuel type in the mining sectors is displayed in Figure 9. The composition by fuel type varies across the mining sectors. Most natural gas was consumed in the oil and gas extraction sector while petroleum-based fuels were the major energy source in coal mining, stone mining, and support activities for oil and gas operations sectors.

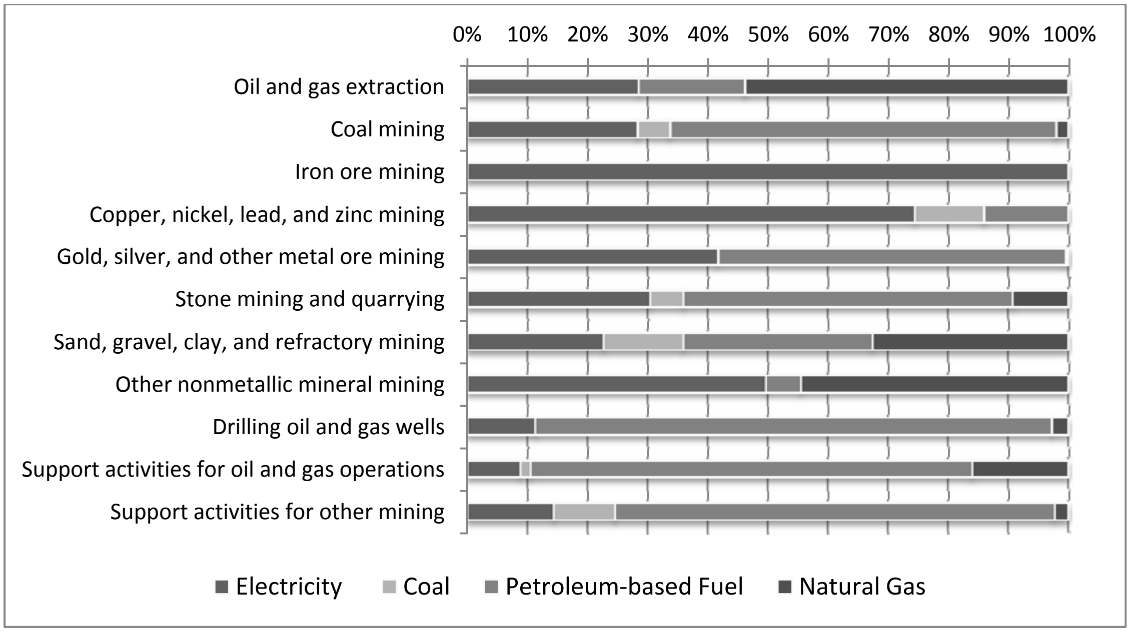

Figure 9: Composition of Energy Consumption by Fuel Type in the Mining Sector

Source: Economic Census 2007 (Sector 21: Mining/ Subject Series: Materials Summary/ Selected Supplies)

Manufacturing Sector: More than two-thirds of industry sectors in IMPLAN are manufacturing sectors, an important segment in the construction of the energy use inventory. The major data sources are MECS 2006 and Economic Census 2007.[25] Detailed fuel consumption data are available in the MECS, including primary fuels such as distillate fuel oil, residual fuel oil, natural gas, and coal, as well as by-product fuels such as coke oven gas (mainly consumed in iron and steel mills), waste gas (mainly consumed in petroleum refineries), petroleum coke (mainly consumed in petroleum refineries), pulping liquor (mainly consumed in pulp, paper, and paperboard mills), wood chips (mainly consumed in pulp, paper, and paperboard mills), and other waste oil and materials. In this research, non-primary fuels are grouped and labeled as "by-product fuel & others." Non-primary energy consumption covers about 35% of total

[25] The MECS conducts the census every three years with a sample of approximately 15,000 establishments in the North America Industry Classification System (NAICS) 31 -33.

manufacturing energy consumption. Figure 10 displays the composition by fuel type consumed in the manufacturing sector. While natural gas (43%) and electricity (19%) comprise the main energy sources in the manufacturing sector, petroleum-based fuel occupies a relatively small portion.

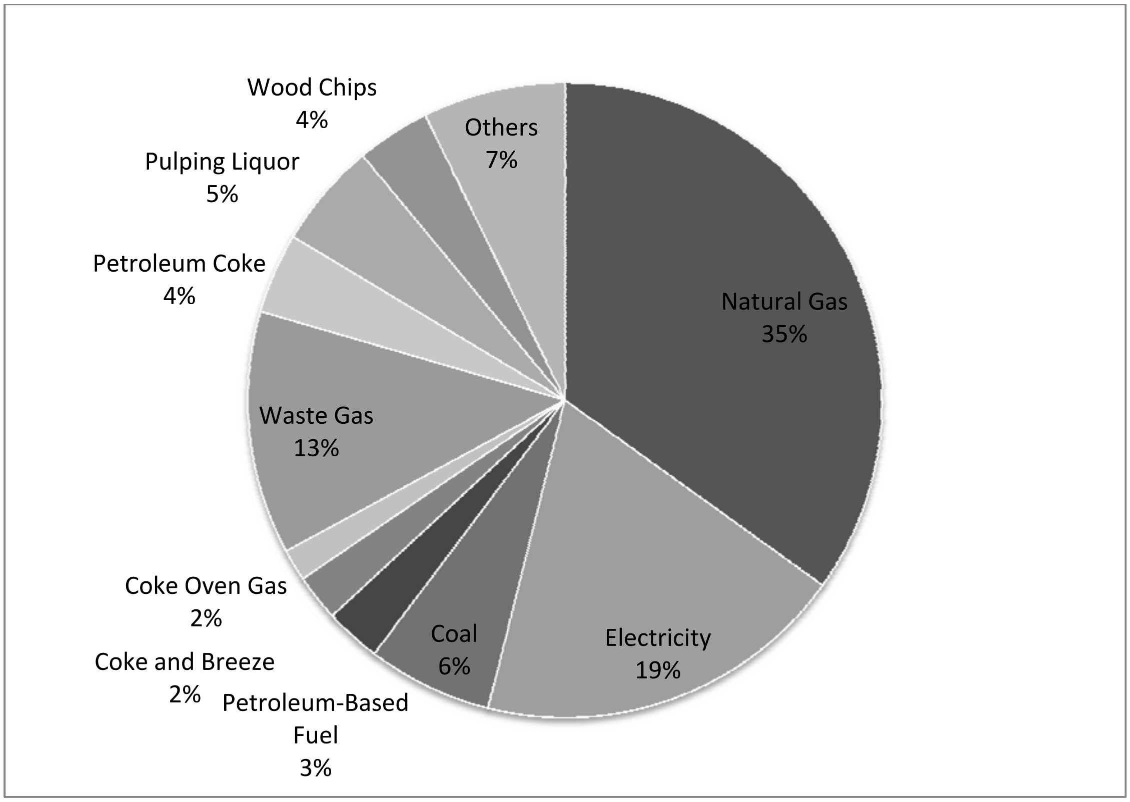

Figure 10: Composition of Energy Consumption by Fuel Type in All Manufacturing Sectors

Source: Manufacturing Energy Consumption Survey 2006

While the MECS details fuel types, the industry classification of the MECS is not as disaggregated as the industry classification of IMPLAN. Hence, we need to allocate the fuel consumption of an aggregated sector in MECS 2006 into the disaggregated IMPLAN sectors. For the allocation of energy consumption, the use table of IMPLAN is utilized. The row vector of the use table indicates the economic value of a commodity that the industry sectors purchase. The use table includes energy commodities. This research takes into account four energy commodities and corresponding energy-producing sectors in IMPLAN: coal mining (IMPLAN 20), power generation and supply (IMPLAN 30), natural gas distribution (IMPLAN 31), and petroleum refineries (IMPLAN 142). The row vectors of these energy-producing sectors are assumed to represent the economic value of energy commodities consumed in each industry sector: coal, electricity, natural gas, and petroleum-based fuel. Using this information, we can calculate the ratio of energy use in dollar value among the disaggregated sectors for the allocation of energy use information in an aggregated sector.

For example, according to the MECS, the textile mill sector, NAICS 313, consumed 46 trillion Btu of natural gas in 2006. The textile mill sector in the MECS is equivalent to seven disaggregated sub-textile mill sectors in IMPLAN (IMPLAN 92-98). The use table indicated the dollar value of natural gas purchased from the natural gas distribution sector, IMPLAN code 31, by seven sub-textile mill sectors. The share of natural gas purchased among the seven sub-textile mill sectors is calculated, and this ratio is applied to allocate 46 trillion Btu of natural gas use of a textile mill into seven sub-textile mill sectors.

Through this allocation procedure, the energy use of five types of fuels in the manufacturing sectors is estimated. Table 15 illustrates the energy consumption patterns of major energy-intensive manufacturing sectors aggregated with three- or four-digit NAICS. Petroleum and coal products, basic chemicals, pulp paper and paperboard, iron and steel mills, and food manufacturing are the top energy-consumption manufacturing sectors, and the portion of by-product fuel use is relatively high in these sectors.

Table 15: Energy Use of the Top Energy-Intensive Manufacturing Sectors in 2006

Manufacturing Sectors	Electricity (TJ)	Coal (TJ)	Petroleum-Based Fuel (TJ)	Natural Gas (TJ)	By-Product Fuel &Others (TJ)
Petroleum and coal products	48,669	55,873	118,070	895,016	2,367,733
Basic chemical	98,508	119,235	22,240	766,479	929,681
Pulp paper and paperboard	57,866	226,653	107,528	355,265	1,256,606
Iron and steel mills	108,470	16,867	25,301	494,420	650,441
Food manufacturing	80,153	154,967	47,439	669,417	113,854
Nonmetallic mineral product	45,860	337,344	40,060	483,878	149,696
Resin rubber artificial fiber	32,564	27,409	10,543	417,277	179,382
Wood products	25,056	2,108	25,301	90,661	255,116
Plastics and rubber products	60,998	10,542	17,921	133,883	0
Converted paper product	15,275	6,325	7,379	144,425	117,016
Motor vehicle body trailer and parts	29,055	1,698	7,583	145,656	8,201
Agriculture chemical	7,804	3,188	1,419	159,150	5,008
Textile mill	16,160	33,734	2,108	68,523	12,650
Hardware/machine shop/coating, etc.	17,583	0	2,989	101,990	0
Foundries	16,233	0	2,108	75,902	27,409
Nonferrous metal production	13,551	4,217	2,108	45,331	9,488

Utility Sector: The electric power generation sector is the most energy-intensive sector. In IMPLAN, three industry sectors relate to electricity generation: electricity power generation, transmission, and distribution (IMPLAN 30), federal electric utilities (IMPLAN 495), and state and local government electric utilities (IMPLAN 498). The U.S. EIA's Annual Energy Review 2006 estimates total fuel consumption for electric power generation sectors for that year: 20,463 trillion Btu of coal, 6,375 trillion Btu of natural gas, and 491 trillion Btu of petroleum-based fuel. These amounts are allocated to three electricity generation sectors following the same allocation procedure used in the manufacturing sector.

Transportation Sector: Estimating energy use in the transportation sectors produces relatively large uncertainty in terms of data sources and allocation procedures. The *Transportation Energy Data Book: Edition 27,* published by the U.S. Department of Energy, is a primary source of energy use information for the transportation sectors. It estimates the annual domestic consumption of transportation energy by transport mode and fuel type. Since the data book does not compile energy use data by industrial sector, we need to approximately allocate energy use by transport modes into industry sectors. For example, jet fuel consumption by air transport modes is allocated to the air transportation sector, IMPLAN 391, and diesel fuel consumption by medium and heavy truck modes is allocated to the truck transportation sector, IMPLAN 394. Whereas these allocations seem intuitively reasonable, allocating some types of fuel use such as that for light trucks can be more challenging. Such light trucks can be utilized for either personal or business purposes. If used for personal purposes, it should be excluded in an estimation of industrial energy use. Because it is unclear how much energy by light truck is consumed for only business purposes, the energy use of light trucks is excluded in the allocation procedure.

Another source of uncertainty relates to whether transportation activity is either out-sourced or operated by in-house transport units. For example, some large retail chains establish logistics systems operated by their own vehicles. In this case, transportation-related fuel consumption is attributed to the retail sector, not to the truck transportation sector. If it is allocated to the truck transportation sector, energy use of this sector is disproportionate to revenue from a truck transportation sector. The Transportation Satellite Accounts (2011) by the U.S. Department of Transportation addressed this issue in the IO model by estimating the size of the economic activity of in-house transportation and for-hire transportation. For example, the industry output of for-hire and in-house truck transportation is $169,397 million and $334,092 million, respectively. Only 33.6% of truck transportation originates in transportation businesses. Thus, total transportation energy consumption estimated in the *Transportation Energy Data Book* needs to be divided into two groups: for-fire and in-house transportation. Thus, the output ratios of in-house to for-hire transportations are utilized for allocation. The percentage of output of for-hire transportation is 87.2% for air, 97.6% for rail, 84.4% for water, and 33.6% for truck transportation. Only for-hire transportation energy use is allocated to the industry transportation sectors. The share of energy consumption by the transportation sectors is displayed in Figure 11.

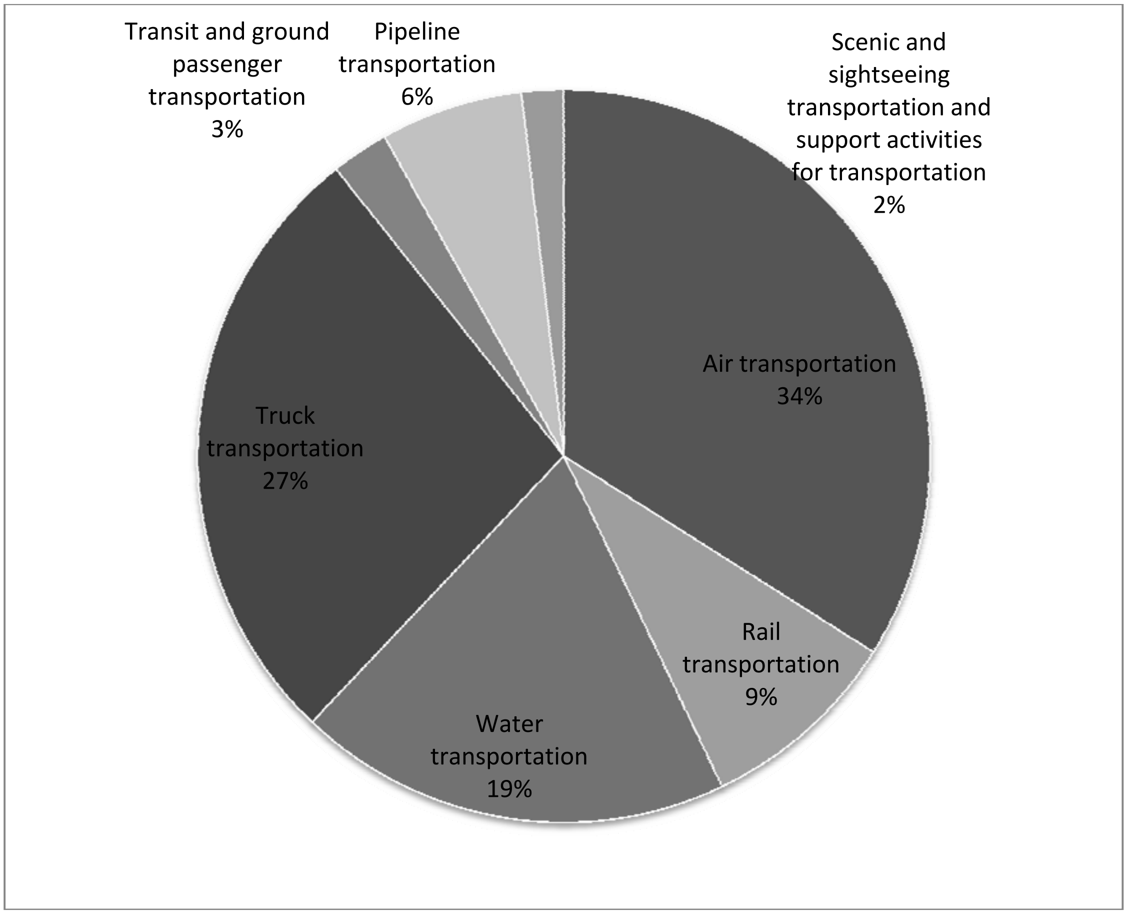

Figure 11: Share of Energy Consumption in the Transportation Sector
Source: Transportation Energy Data Book 27 Edition (Table 2.5)

Federal Government: Annual Energy Review 2006 provides information pertaining to the energy use of federal government agencies such as defense, the postal service, transportation, the general service administration, the justice department, and agriculture etc. It includes several fuel types such as coal, natural gas, aviation gasoline, distillate and residual fuel oil, jet fuel, motor gasoline, LPG, and electricity. IMPLAN contains three sectors of the federal government that match federal government agencies in Annual Energy Review 2006: postal service (IMPLAN 398), federal military (IMPLAN 505), and federal non-military (IMPLAN 506). The energy consumption data from each federal government agency in the Annual Energy Review2006 are allocated to associated IMPLAN sectors.

Service and Other Sectors: No useful statistics are available for forestry and fishing (IMPLAN 14-18), construction (IMPLAN 33-45), service (IMPLAN 390/399-494), and other local government sectors. For these sectors, the use table in the national IO model for 2006 is utilized for estimating energy use. As noted in the allocation for the manufacturing sector, this research considers four energy-producing sectors: coal of the coal mining sector (IMPLAN 20), electricity of the power generation and supply sector (IMPLAN 30), natural gas of the natural gas distribution sector (IMPLAN 31), and petroleum-based fuel of the petroleum refineries sector (IMPLAN 142). It assumes that the purchase of input commodities from these sectors recorded in the use table represents the energy commodities of coal, electricity, natural gas, and petroleum-based fuel consumed in the service, construction, and forestry sectors. The dollar values of input commodities purchased from the coal mining, power generation and supply, natural distribution, and petroleum refineries sectors are converted into energy units. For example, the legal service sector (IMPLAN 437) purchased $87 million of commodities from the natural gas distribution sector. This dollar value is converted into the physical quantity of energy using the average natural gas price in 2006, and then the physical quantity of natural gas is converted into energy units.

Figure 12 shows the composition of energy consumption in the service sectors aggregated into two-digit NAICS. Electricity and natural gas mainly used for lighting and heating are common energy sources in most of the service sectors: finance insurance (65.3%), real estate (74.6%), professional and technical service (92.8%), and art entertainment and recreation (89.2%). Petroleum-based fuel is a primary source of energy for several sectors such as wholesale and retail trade, administrative and waste management, and health care and social assistant sectors. These sectors mainly pertain to the industrial activities of in-house transportation.

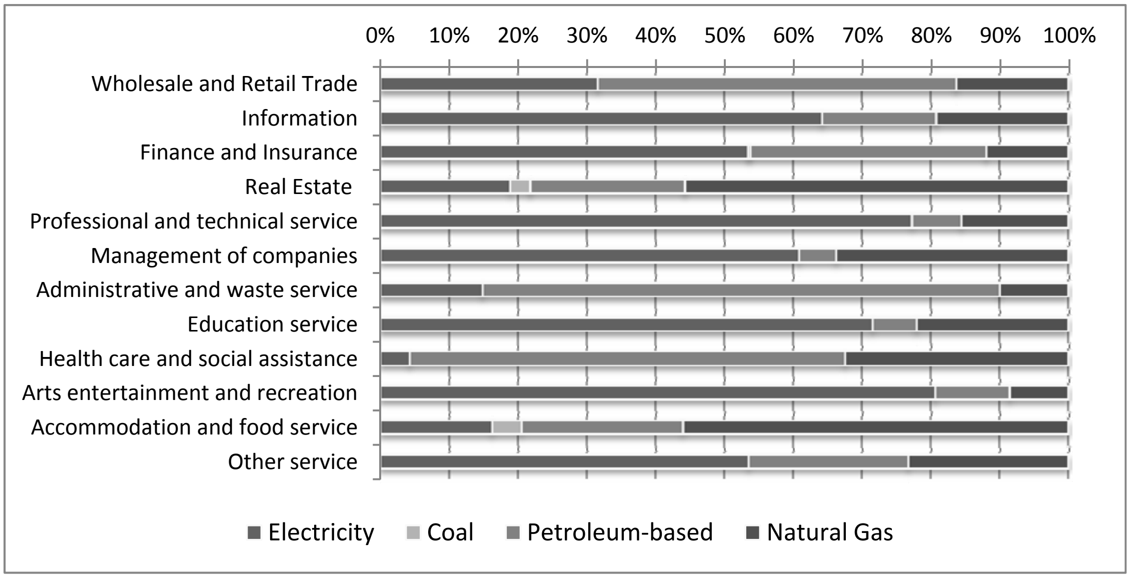

Figure 12: Composition of Energy Consumption in the Service Sector

Through the disaggregation, conversion, and allocation procedures of each sectoral group, energy use of electricity, coal, natural gas, petroleum-based fuel, and by-product fuel & others are estimated for 509 IMPLAN sectors. Total energy use of the aggregated industry sectors is summarized in Figures 13 and 14. The utility sectors consumed 44.3% of all types of fuels while the output share of the utility sectors was only 1.8%. The manufacturing sectors accounted for 21.6% of energy consumption and 25.6% of total output. Transportation sectors consumed 10.8% of total fuels with 3.1% of total output. The output share of the service sectors, which consumed only 9.7% in total, was 40.3%.

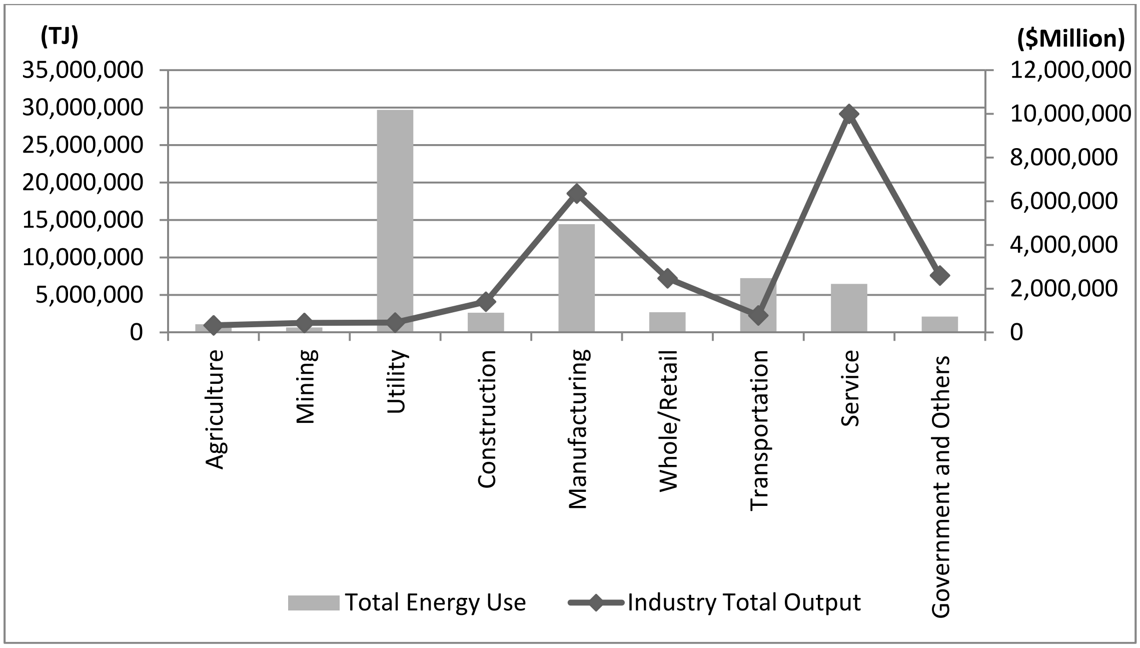

Figure 13: Total Energy Use and Output by Industry Sector

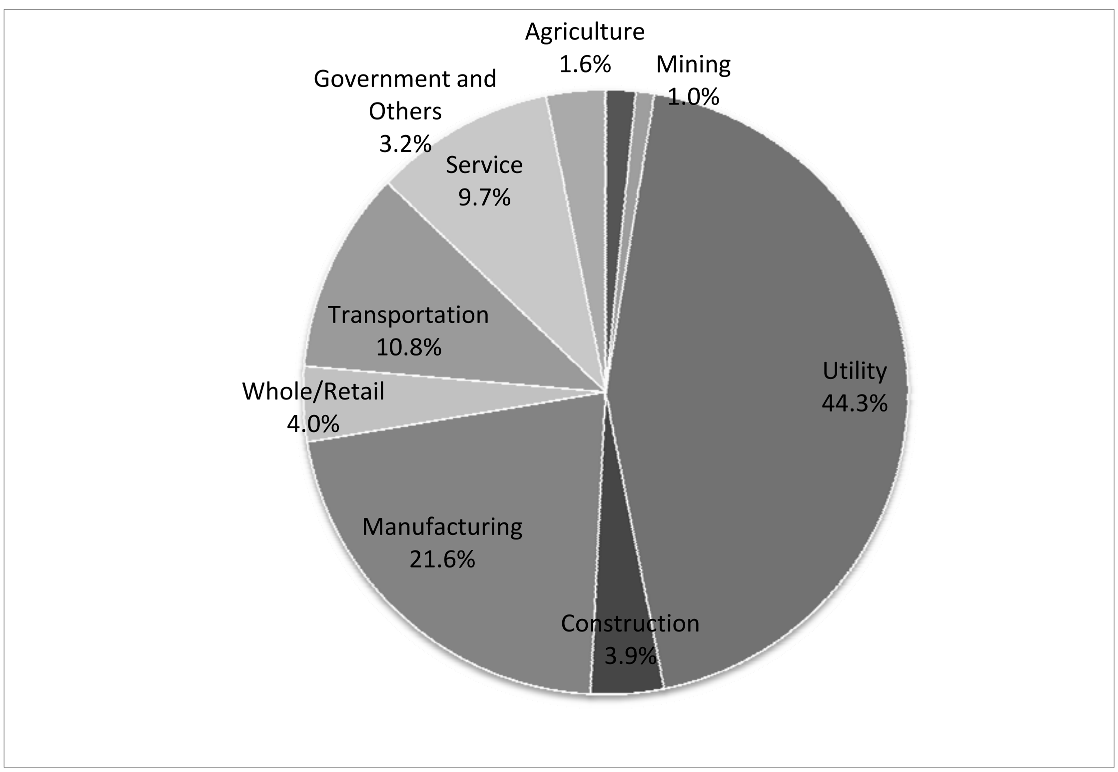

Figure 14: Share of Total Energy Consumption by Industry Sector

After the estimates of industry energy use by fuel type are updated, energy use coefficients are calculated by dividing industry energy use by the total industry output. The energy use coefficients of selective sectors are displayed in Table 16. The table shows that the power generation and supply sector (IMPLAN 30) is the most energy-intensive sector, followed by pulp mills (IMPLAN 124), cement manufacturing (IMPLAN 191), paper and paperboard mill (IMPLAN 125), petroleum refineries (IMPLAN 142), iron and steel mills (IMPLAN 203), and alumina refining (IMPLAN 208).

Table 16: National Energy Use Coefficients by Fuel Type of Selective Industry Sectors

IMPLAN code	Industry Description	Non-Fossil Electricity (Mkwh/$M)	Coal (TJ/$M)	Petroleum-Based Fuel (TJ/$M)	Natural Gas (TJ/$M)	By-Product Fuel & Others (TJ/$M)
24	Stone mining and quarrying	0.182	0.387	3.819	0.650	0.000
25	Sand, gravel, clay, and refractory mining	0.154	1.051	2.498	2.582	0.000
30	Power generation and supply	0.230	61.069	1.900	24.668	0.000
31	Natural gas distribution	0.004	0.000	0.109	4.709	0.000
92	Fiber, yarn, and thread mills	0.175	0.256	0.018	0.504	0.273
99	Carpet and rug mills	0.023	0.093	0.029	1.395	0.000
112	Sawmills	0.070	0.000	0.260	0.390	3.057
116	Engineered wood member and truss manufacturing	0.027	0.095	0.124	1.247	2.368
124	Pulp mills	0.145	1.448	3.619	3.137	38.608
125	Paper and paperboard mills	0.218	2.792	1.162	4.328	13.785
142	Petroleum refineries	0.023	0.002	0.156	1.397	3.942
152	Plastics material and resin manufacturing	0.099	0.138	0.088	4.183	1.796
153	Synthetic rubber manufacturing	0.049	0.580	0.000	2.511	0.580
155	Noncellulosic organic fiber manufacturing	0.089	0.758	0.253	2.862	1.768
191	Cement manufacturing	0.400	22.783	0.388	1.939	7.950
192	Ready-mix concrete manufacturing	0.023	0.060	0.270	1.450	0.150
197	Gypsum product manufacturing	0.095	0.000	0.125	9.534	0.251
203	Iron and steel mills	0.214	0.179	0.281	4.163	7.777
208	Alumina refining	0.042	0.000	0.302	13.658	0.098
394	Truck transportation	0.000	0.000	7.267	0.000	0.000
460	Waste management and remediation services	0.053	0.000	9.113	1.211	0.000

Verifying the accuracy of the estimates of the energy use inventory is an important task. Jensen (1980) discussed the concept of accuracy of the IO model. He posited the partitive and holistic accuracy. Partitive accuracy implies cell-by-cell accuracy in a statistical sense. The IO table is constructed using various survey data on individual firms and their transactions. If the IO tables reflect real transactions, it should have a high-level of partitive accuracy. However, even well-designed survey forms suffer from various sources of errors such as hiding information, inadequate training of observers, and incorrect definition and classification. Jensen contended that partitive accuracy is difficult to test, particularly in regional IO tables with limited data sources and resources.

Different from partitive accuracy, holistic accuracy tests whether the IO table represents well the main characteristics of the economy in a descriptive sense in terms of size and structure. Jensen suggested that holistic accuracy is a modest and tenable goal in the context of regional IO modeling. He described holistic accuracy as minimization of distance between calculated and true tables. However, since true value is not accessible, we may compare the IO table and other statistics that are not used in the construction of the IO table. The practical approach of holistic accuracy is still problematic in that comparable datasets that are not used in construction of the regional IO table are scarce and we need to establish the criterion of an acceptable error limit.

The holistic accuracy approach is applied for testing the accuracy of estimates of the energy use inventory in a highly limited case because the few data that are not used in estimation remain for comparison. The estimates of total energy use by fuel type are compared to other government statistics from the U.S. EIA's State Energy Consumption

Estimates. The comparison, which is quite crude, is displayed in Table 17. Differences in coal, petroleum-based fuel and other liquids, and the sum of fuels are less than 5%. Natural gas consumption appears to be underestimated on a relatively large scale. The natural gas consumption of the manufacturing sectors estimated from the MECS is lower than that from the report in the State Energy Consumption Estimates. One possible reason for this difference is that nonfuel (feedstock) energy use[26] in the MECS is not included in the estimation procedure.

Table 17: Comparison Between Total Energy Use from EIA Data and that from Estimates of the National Energy Use Inventory

Fuel Type	U.S. EIA Data (Trillion Btu)	Estimates of Energy Use Inventory (Trillion Btu)	Ratio
Coal	22,439	21,579	96.2%
Natural Gas	17,795	15,929	89.5%
Petroleum-Based Fuel and Other Liquids	22,399	23,145*	103.3%
Sum of Fuels	62,633	60,652	96.8%

* By-product fuels are included in the petroleum-based fuel.
Sources: State Energy Data System

5.1.2. Greenhouse Gas Emissions

On the basis of estimates of the national industry energy use inventory, this research computes the GHG inventory and GHG emissions coefficients connected to the national IO tables. This IO-based GHG inventory is a subset of the U.S. GHG inventory reported by the U.S. EPA (2011). The report of the U.S GHG inventory, which is

[26] Nonfuel (feedstock) energy use in the MECS is defined as "energy used for purposes other than for heat power, and electricity generation (called nonfuel purposes)" U.S. EIA website accessed on June 14, 2012 at http://www.eia.gov/emeu/mecs/mecs98/datatables/nonfueldef.html

consistent with recommendations of the Intergovernmental Panel on Climate Change (IPCC), includes five sources of GHG emissions: energy, industrial process and product use, agriculture, land-use, and waste. The types of GHG are CO_2, CH_4, N_2O, HFCs, PFCs, and SF_6.

This research focuses on CO_2 emissions generated from fossil fuel combustion by industry sector. It excludes GHG emissions from the energy consumption for residential purposes such as the heating and cooling of residential buildings and the use of personal vehicles In addition, this study does not include GHG emissions from non-fossil fuel sources because of the difficulty in allocating GHG emissions to industry sectors and lack of information for region-specific GHG emission factors. For example, in the case of GHG emissions from land use changes or agricultural activities such as soil and manure management, the allocation of emissions to a specific industry sector in the IO classification is challenging. The share of U.S. GHG emissions generated from fossil fuel combustion in total (without sink) ranged from 78.9% in 2006 to 83.0% in 2009. The categories of the GHG inventory by emission source of the U.S. EPA report and those of the IO-based model are compared in Table 18.

Table 18: Comparison between the Covered GHG Emission Sources of the U.S. GHG Inventory by the U.S. EPA and those of the IO-Based GHG Inventory

Sources of GHG		U.S GHG Inventory (EPA)	IO-Based GHG Inventory
Energy	Residential	O	×
	Industrial	O	O
	Commercial	O	O
	Transportation	O	Δ
	Electric Power	O	O
Industrial Process and Product Use		O	×
Agriculture		O	×
Land-use		O	×
Waste		O	×

Note: O include; × exclude; Δ partially include

GHG emissions from fossil fuel combustion by industry sector are calculated by multiplying the estimate of industry fuel consumption by the GHG emission factor. The emission factors are displayed in Table 19. The total estimate of GHG emissions of the IO-based approach is 4,642 Tg CO_2 Eq. in 2006. Per capita GHG emissions are 15.6 ton CO_2 Eq. In a U.S. EPA report, CO_2 emissions from fossil fuel combustion were 5,653.1 Tg CO_2 Eq. in 2006. If GHG emissions from residential use and 95% of the GHG emissions from motor gasoline transportation are subtracted from total CO_2 emissions, CO_2 emissions are 4,212.3 Tg CO_2 Eq. The difference between the estimates in the U.S.EPA report and the IO-based GHG inventory is roughly 10%.[27]

[27] The exact proportion of motor gasoline use for non-business purposes is unknown. If the 80% of GHG emissions from motor gasoline transportation is excluded, the difference decreases 5%.

Table 19: CO_2 Emission Factors by Fuel Type

Fuel Type	Fuel Consumption (TBtu)	Emissions from Energy Use (Tg CO_2 Eq.)	Emission Factor (Ton CO_2 Eq./TBtu)
Electric Power Coal	20,462	1,954	95,480
Natural Gas (Industry)	7,125	378	53,011
Jet Fuel(Transportation)	2,347	170	72,217
Distillate Fuel Oil (Industry)	1,199	89	73,954
LPG (Commercial)	123	8	61,688
Residual Fuel (Industry)	176	13	74,830
Petroleum Coke (Industry)	683	70	102,125

Source: 2011 U.S. Greenhouse Gas Inventory Report Annex 2 (Table A-16)

The share of GHG emissions by industry sector is displayed in Figure 15. The utility sector emitted more than half of total GHG emissions (51.0%). Manufacturing and transportation sectors are responsible for 19.0% and 10.6% of GHG emissions, respectively. The Wholesale and retail, construction, agriculture, mining, and government sectors contribute to less than 5% of total GHG emissions.

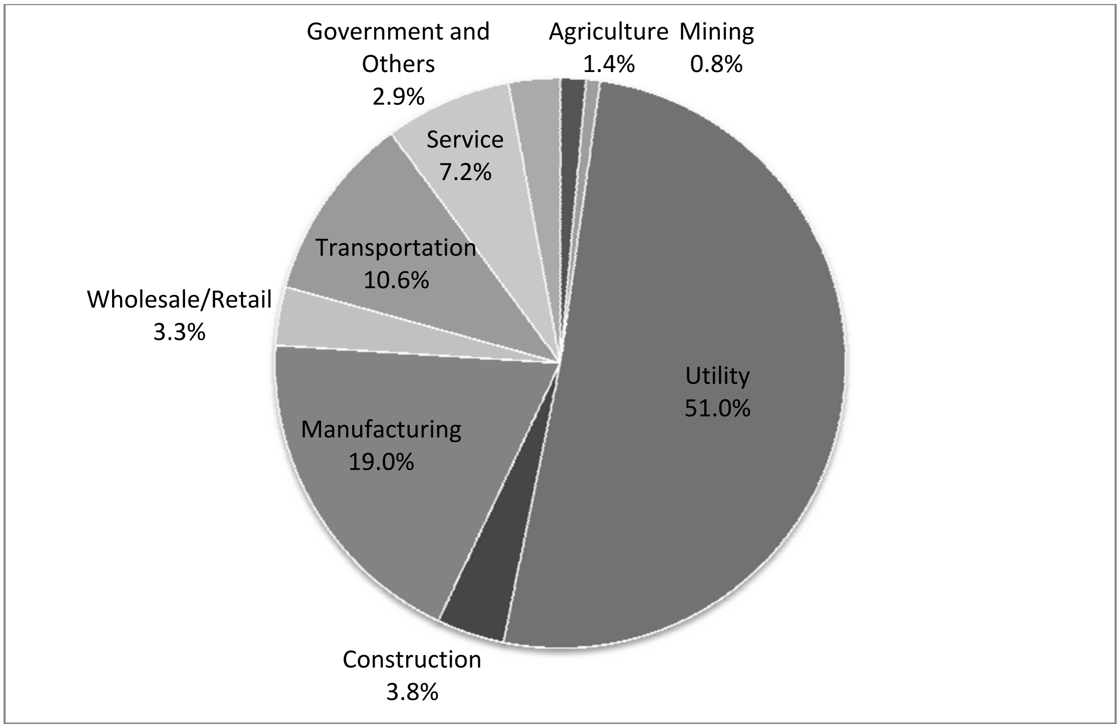

Figure 15: Share of GHG Emissions by Aggregated Industry Sectors

Finally, GHG emission coefficients by industry sector are calculated by dividing the total industry GHG emissions by the dollar value of output. The selective cases of GHG emission coefficients (tons of GHG emissions per million dollars) are displayed in Table 20. As expected, the energy-intensive sectors such as power generation and supply (IMPALN 30), pulp mills (IMPLAN 124), cement manufacturing (IMPALN 191), and lime manufacturing (IMPALN 196) are also heavy emitters of GHGs.

Table 20: Selective Cases of GHG Emission Coefficients by Industry Sector

IMPLAN code	Industry Description	GHG Emission Coefficients (Ton CO_2 Eq. per \$million)
24	Stone mining and quarrying	334.6
25	Sand, gravel, clay, and refractory mining	399.2
30	Power generation and supply	6,904.8
92	Fiber, yarn, and thread mills	68.6
99	Carpet and rug mills	80.2
124	Pulp mills	3,807.5
125	Paper and paperboard mills	1,683.8
142	Petroleum refineries	364.5
152	Plastics materials and resin manufacturing	346.0
155	Noncellulosic organic fiber manufacturing	350.1
178	Foam product manufacturing	79.0
191	Cement manufacturing	2,867.3
192	Ready-mix concrete manufacturing	107.2
196	Lime manufacturing	6,373.8
197	Gypsum product manufacturing	505.7
203	Iron and steel mills	933.7
208	Alumina refining	714.8
394	Truck transportation	507.8
460	Waste management and remediation services	700.6

5.2. Regional Environmental Inventory

The previous section showed that the abundant U.S. statistics on energy use have allowed us to construct the national-level energy use and GHG inventory and coefficients by industry sector through disaggregation and allocation procedures. This section examines the regionalization method of energy use and GHG emission coefficients with limited region-specific information. It briefly explores the sources of data utilized for regionalization of environmental coefficients and then describes the practical steps to establishing regional energy use and GHG emission coefficients for three metropolitan areas.

5.2.1. Energy Use

The construction of the coefficients for regionalized energy use raises fundamental questions: Why do industrial energy use patterns vary from region to region? And how do they differ? Differences in the age and applied technology of a facility, the climate, the network of roads, the modes of transportation, and the issue of the product-mix can be contributing factors to variations in the patterns of energy use across region. Regional energy use coefficients are expected to capture variations. The construction of a regional energy use inventory necessitates regional-specific energy use statistics. Sub-national energy use data in the U.S., however, are relatively limited compared to national-level energy use data, and as spatial units become finer, available energy use data become scarcer. In particular, metropolitan-level industry-specific energy use statistics are extremely rare. Therefore, although the spatial unit of analysis is a metropolitan area, the regionalization of energy use coefficients relies on statistical data containing wider spatial units such as state and census region.

The major data sources for the regionalization of energy use coefficients are State Energy Consumption Estimates for the electric power generation sector, the 2007 Commodity Flow Survey for the truck transportation sector, the MECS 2006 by the Census Region for the manufacturing sector, and the use table of the metropolitan-specific IMPLAN IO model for the construction, service, and other sectors. Relevant data sources and covered fuel types are summarized in Table 21. Energy use coefficients of agriculture and mining sectors were not regionalized due to lack of relevant sub-national data. Agriculture and mining, which comprise less than 1% of the total industry output of all sectors in the three metropolitan cases, are typically small industry sectors in the metropolitan economy.

Table 21: Data Sources for Regionalizing Energy Use Coefficients

Sectoral Group	Sources	Type of Energy Consumption (Units)	IMPLAN Sector Code
Utility	State Energy Consumption Estimate	Energy Consumption for Electricity Generation: coal/natural gas/petroleum (in trillions of Btu)	30/495/498
Manufacturing	Manufacturing Energy Consumption Survey 2006: Energy Consumption by Census Region (Northeast, Midwest, South, West)	13 Categories of energy: electricity, residual fuel oil, distillate fuel oil, natural gas, coal, coke and breeze, coke oven gas, waste gas, petroleum gas, pulping liquor, wood chips, waste oil and materials (in trillions of Btu)	46- 389
Transportation	2007 Commodity Flow Survey	For-hire truck ton-miles	394
Construction, Service, and Government	IMPLAN 2006 Metropolitan Use Table	Commodity purchase from energy producing sectors ($ millions) Coal: coal mining Electricity: power generation and supplies Natural gas: natural gas distribution Petroleum-based: petroleum refineries	33 -45 (Construction) 390/399-494 (Service) 495- 509 (Government)

Utility Sector: Types of fuels consumed in the power generation sector significantly differ across states. Table 22 shows the variety of energy sources for electric power generation at a state level. Electricity generation in Georgia heavily relies on fossil fuel combustion (combined fossil fuel: 72.6%) while Washington exploits their plentiful water resources for electricity generation (hydroelectric power: 76.8%). In California, while natural gas is a primary energy source, non-fossil fuel sources occupy a large portion, 56.1%. Information from the state energy use profile for electricity generation in the State Energy Consumption Estimates is used to estimate the state-specific energy use coefficients for the power generation and supply sector. Each set of coefficients is applied to a corresponding metropolitan-level environmental IO model. The state-specific energy use coefficients are displayed in Table 22. The energy use coefficient of coal in Georgia is higher than that of the nation whereas the energy use coefficient of coal in California and Washington is significantly lower than that of the nation. The energy use coefficient of natural gas in California is similar to the national average. The energy use coefficients of natural gas and petroleum-based fuel for electricity generation in Washington are lower than the national averages.

Table 22: Share of Energy Sources in Electric Power Generation by State in 2006

	California	Georgia	Washington	National
Share of Energy Sources				
Coal	1.1%	64.9%	6.3%	51.9%
Natural Gas	41.6%	7.6%	5.7%	16.2%
Petroleum-based Fuel	1.2%	0.1%	0.0%	1.6%
Nuclear	17.4%	25.5%	9.2%	20.8%
Hydroelectric	24.9%	1.9%	76.8%	7.2%
Others	13.8%	0.0%	2.0%	2.2%
Energy Use Coefficients (TJ/$M)				
Coal	0.6	71.9	7.7	61.1
Natural Gas	23.6	8.4	6.9	24.7
Petroleum-based Fuel	0.7	0.1	0.0	1.9

Sources: State Energy Consumption Estimates, U.S. EIA (2011)

Manufacturing Sector: The sub-national energy use per output in the manufacturing sector may deviate from the national average because of variations in the age and efficiency of installed equipment and because of the product mix issue. MECS 2006, which contains energy use information on the census region level, shows energy use per shipment by fuel type for four census regions. Thus, we can compare the energy use pattern of the manufacturing sector on a census-region scale by calculating the ratio of national energy use per shipment to census region energy use per shipment for each industry sector. And, this ratio is utilized to adjust the national-level energy use coefficients.

For example, according to MECS 2006, the national-level energy use per shipment of the beverage manufacturing sector is 1.1 thousand Btu while the south-census region energy use per shipment of the same sector is 1.3 thousand Btu. Therefore, the ratio of national energy use per shipment to south-census region energy use per shipment is 0.846. The estimated national energy use coefficients of the soft drink and ice manufacturing sector (IMPLAN 85), a sub-sector of the beverage manufacturing sector are adjusted by a ratio of 0.946 for the south-census region. In this way, the energy use coefficients in the manufacturing sector are regionalized, and the census-region energy use coefficients apply to corresponding metropolitan-level environmental IO models.

Transportation Sector: The different composition of the types of transportation vehicles and the variations in shipping distances may contribute to diverse energy use patterns in transportation across regions. Since no useful data are available for sub-national energy use in detail transportation sectors, the state-level energy use is directly estimated by considering the differences of shipping distances and vehicle types. However, only a truck transportation sector is regionalized owing to lack of relevant information about the composition of vehicle types in other transportation sector at the state level.

The 2007 Commodity Flow Survey estimated the total ton-miles of for-hire trucks in a case of the state of origin. To calculate the total energy use of the truck transportation sector, the information pertaining to type of vehicle utilized in transporting is required. Trucks are classified by their gross vehicle weight rating (GVWR), and their fuel efficiency widely differs according to the GVWR. Typical ton-mile per gallon (ton-mpg) in Class 4 (medium duty truck) is 42 while that of Class 8 (heavy duty) is 155. If heavy duty trucks make up a larger portion of a state's truck vehicles, the average energy use per shipment will be lower. Information about the number of truck vehicles by class at the state level is available in the 2002 Vehicle Inventory and Use Survey, shown in Table 23.

Using the composition of truck vehicles by class at the state level and typical ton-mpg, the average ton-mpg for truck transportation is calculated for three states. Then, the total energy use of truck transportation is calculated by multiplying the total ton-miles of a for-hire truck by the average ton-mpg. Finally, the state-level energy use coefficients of the truck transportation sector are calculated by dividing estimates of total energy use by the output of the truck transportation sector: 7.2 TJ per million dollars in California, 8.8 TJ per million dollars in Georgia, and 9.4TJper million dollars in Washington. Those state-level coefficients are used in corresponding metropolitan cases respectively.

Table 23: Vehicle Classification and Number of Trucks by Class and State

Class	GVWR	Typical Ton-mpg	Number of Truck (Thousands) California	Georgia	Washington
2b	8,501-10,000	26	317.1	81.8	64.4
3	10,001-14,000	30	72.5	44.7	44.6
4	14,001-16,000	42	38.2	12.1	5.9
5	16,001-19,500	39	22	11.8	5
6	19,501-26,000	49	72.4	34.6	17.4
7	26,001-33,000	55	40.4	11.3	7.2
8	33,001-80,000	155	159.4	38.5	34.1

Source: U.S.EPA and U.S. Department of Transportation (2011)-Typical Ton-mpg; Vehicle Inventory and Use Survey 2002-Number of Truck

Service and Other Sectors: The energy use in the service sectors is mainly associated with the cooling, heating, and lighting of buildings and in-house transportation. Although building-related energy use differs across regions owing to climate and building energy efficiency, no comprehensive building energy use dataset pertaining to disaggregated IO industry classification is available. The regionalization of service and construction sectors relies on metropolitan-level use tables. The procedure for estimation is the same as that for the national energy use inventory.

By applying the regionalization procedure by sectoral group, this study estimates the metropolitan-level energy use inventory and coefficients by industry sector. Energy use by aggregated industry sector for three metropolitan areas is summarized in Table 24. The table shows that total energy consumption in the Atlanta metropolitan area is higher than that in the other two metropolitan areas, which results from its heavy dependence of electric power generation on fossil fuel combustion. In addition, energy use shares by industry sector considerably differ across metropolitan areas. The share of the energy use ratio in manufacturing sectors in the San Francisco metropolitan area is notably larger than it is in others. The large sizes of the petroleum refineries in the San Francisco metropolitan area contribute to intensive energy use in the manufacturing sector.

Table 24: Energy Consumption by Industry Sector in Metropolitan Areas

Industry Sectors	Atlanta		San Francisco		Seattle	
	Energy Use (TBtu)	Share	Energy Use (TBtu)	Share	Energy Use (TBtu)	Share
Agriculture	1	0.1%	2	0.2%	8	1.4%
Mining	1	0.1%	2	0.2%	1	0.3%
Utility	607	52.2%	95	10.6%	87	16.1%
Construction	45	3.8%	38	4.2%	40	7.4%
Manufacturing	94	8.1%	362	40.4%	66	12.2%
Whole/Retail	55	4.7%	49	5.5%	48	8.9%
Transportation	235	20.2%	163	18.2%	152	28.3%
Service	112	9.7%	152	17.0%	115	21.3%
Government	13	1.1%	32	3.6%	22	4.1%
Total	1,164		894		538	

5.2.2. Greenhouse Gas Emissions

On the basis of the estimates of industry energy use on the metropolitan level, industry GHG emissions generated from energy consumption are computed for the three metropolitan cases. GHG emissions are calculated by multiplying the quantity of fuel consumed in each industry sector by the CO_2 emissions factor, shown in Table 19. The estimates of GHG emissions by aggregated industry sector in the three metropolitan areas are displayed in Table 25. As expected, the levels of total GHG emissions significantly differ across the three cases: the Atlanta metropolitan area (89.6 Tg); the San Francisco metropolitan area (52.4 Tg); and the Seattle metropolitan area (33.2 Tg).

When they are converted to GHG emissions per capita, the variation is still significant: the Atlanta metropolitan area (20.1 tons per capita); the San Francisco metropolitan area (12.5 tons per capita); and the Seattle metropolitan area (9.5 tons per capita). The dependency of electric power generation on coal in the Atlanta metropolitan area explains most of differences in GHG emissions.

Table 25: GHG Emissions by Industry Sector in Three Metropolitan Areas

Industry Sectors	Atlanta		San Francisco		Seattle	
	GHG Emission (Tg)	Share	GHG Emission (Tg)	Share	GHG Emission (Tg)	Share
Agriculture	0.1	0.1%	0.1	0.2%	0.5	1.6%
Mining	0.1	0.1%	0.1	0.2%	0.1	0.3%
Utility	54.5	60.2%	5.1	9.7%	6.4	19.1%
Construction	3.2	3.5%	2.7	5.1%	2.8	8.2%
Manufacturing	5.6	6.2%	24.2	46.0%	4.2	12.6%
Whole/Retail	3.4	3.8%	2.7	5.1%	2.4	7.1%
Transportation	17.0	18.8%	8.6	16.4%	11.1	33.1%
Service	6.1	6.7%	6.9	13.2%	4.6	13.8%
Government and Others	0.6	0.6%	2.2	4.1%	1.4	4.2%
Total	90.4		52.6		33.6	

CHAPTER 6. REGIONAL GREENHOUSE GAS EMISSIONS INVENTORY USING REGIONAL ENVIRONMENTAL INPUT-OUTPUT MODELING

The previous chapters presented the regional environmental IO modeling frameworks and procedures to estimate regional GHG emission coefficients for three metropolitan areas. The intention of this chapter is to compute the amount of GHG emissions for which each metropolitan economy is responsible with respective to the four principles of environmental responsibility of a regional economy using the single- and two-region approaches. Then, it will present the results of the estimation of GHG emissions that a metropolitan economy should take responsibility for and discuss the findings and implications of the sub-national GHG emission inventory study.

6.1. Construction of the Regional Environmental Input-Output Modeling Frameworks

The single- and two-region environmental IO models require several datasets: regional technical coefficients for metropolitan and national economies; intra- and inter-regional trade between a metropolitan economy and the rest of the nation; inter-regional and intra-regional input coefficients for metropolitan and national economies; final demand components, including the consumption of regionally-produced products, the consumption of imported products, and the exports of regionally-produced products; and GHG emission coefficients for metropolitan and national economies.

Regional Technical Coefficients: Regional technical coefficients, labeled as gross absorption coefficients, are available in the IMPLAN IO tables. These data are directly obtained from the IMPLAN dataset for the national and three metropolitan economies.

Intra- and Inter-Regional Trade: The information pertaining to intra- and inter-regional commodity trade by industry sector between a metropolitan area and the rest of the nation is essential for two-region environmental IO modeling. Specifically, six datasets are required as shown in Equation 19 in Chapter 4: regionally-supplied products, domestically-imported products, and internationally-imported products to a metropolitan area and the rest of the nation. These data sets can be obtained from IMPLAN.

The IMPLAN dataset not only informed the proportions of regionally-supplied products in the metropolitan economy, which are RPC, from which we can calculate the amount of regionally-supplied products, but also contained the quantity of domestic- and international-imported products in the metropolitan economy. Hence, given information about the amount of regionally-supplied, domestically-imported, and internationally-imported products in the metropolitan economies, we calculated the proportion of geographical area from which the product supplies originated for each metropolitan economy

Regarding the rest of nation, in a two-region setting, the domestic exports of a metropolitan area are equal to domestic imports of the rest of the nation. The number of regionally-supplied products of the entire nation can be calculated from RPC in the national-level IMPLAN data, and the number of internationally-imported products to a nation is directly obtained from the national-level IMPLAN data. If the number of domestic exports of a metropolitan area is subtracted from the number of regionally-supplied products of a nation, the result is the number of regionally-supplied products of the rest of the nation. Finally, if the number of international imports of a metropolitan area is subtracted from that of the international imports of a nation, the result is the number of internationally-imported products of the rest of the nation. Consequently, given the three datasets for the rest of the nation, we also obtained the proportions of the geographical areas from which the product supplies in the rest of the nation originated.

Inter-Regional and Intra-Regional Input Coefficients: Given the information pertaining to the product-supply proportions of the geographical areas for a metropolitan area and the rest of the nation, the inter-regional and intra-regional input coefficients were calculated from Equation 20 in Chapter 4. The regional technical coefficients were row-adjusted by the product-supply proportions of the geographical areas.

Final Demand: The final demand terms consist of the consumption of regionally-supplied products and imported products and the exports of regionally-produced products. IMPLAN provides estimates of those disaggregated final demand components, shown in Table 26. The shares of the exports and the consumption of regionally-produced and imported products are similar in three metropolitan areas. Approximately 40 to 42 percent of final demand is the regional consumption of regionally-produced products while 14 to 16 percent is the regional consumption of imported products. The proportion of exports in the final demand ranged from 41 to 45 percent.

GHG Emission Coefficients: The national- and metropolitan-level GHG emission coefficients constructed in the previous chapter are utilized for this environmental IO model.

Table 26: Final Demand and Trade in Three Metropolitan Areas

Industry Sector	Final Demand Components			Share of Final Demand		
	Regional Consumption			Regional Consumption		
	Regionally Produced	Imported	Export	Regionally Produced	Imported	Export
Atlanta Metropolitan Area						
Agriculture	66	758	153	6.8%	77.6%	15.6%
Mining	2	78	292	0.4%	20.9%	78.7%
Utility	3,873	486	3,077	52.1%	6.5%	41.4%
Construction	23,849	497	45	97.8%	2.0%	0.2%
Manufacturing	12,532	29,016	40,484	15.3%	35.4%	49.4%
Wholesale & Retail	23,828	1,007	22,046	50.8%	2.1%	47.0%
Transportation	2,354	1,178	8,056	20.3%	10.2%	69.5%
Service	90,895	22,621	98,755	42.8%	10.7%	46.5%
Sum	157,399	55,642	172,907	40.8%	14.4%	44.8%
San Francisco Metropolitan Area						
Agriculture	64	951	392	4.5%	67.6%	27.9%
Mining	11	105	258	2.9%	28.2%	68.9%
Utility	2,866	1,995	9,101	20.5%	14.3%	65.2%
Construction	22,087	215	1,925	91.2%	0.9%	7.9%
Manufacturing	22,251	32,329	84,514	16.0%	23.2%	60.8%
Wholesale & Retail	26,792	3,052	5,397	76.0%	8.7%	15.3%
Transportation	2,858	1,360	6,586	26.5%	12.6%	61.0%
Service	122,325	29,862	106,859	47.2%	11.5%	41.3%
Sum	199,253	69,869	215,032	41.2%	14.4%	44.4%
Seattle Metropolitan Area						
Agriculture	120	657	817	7.6%	41.2%	51.3%
Mining	3	52	260	0.9%	16.6%	82.4%
Utility	2,795	637	2,612	46.2%	10.5%	43.2%
Construction	17,054	156	3,174	83.7%	0.8%	15.6%
Manufacturing	15,793	25,067	50,021	17.4%	27.6%	55.0%
Wholesale & Retail	20,144	1,965	9,955	62.8%	6.1%	31.0%
Transportation	2,024	795	5,205	25.2%	9.9%	64.9%
Service	88,508	27,260	69,158	47.9%	14.7%	37.4%
sum	146,442	56,589	141,201	42.5%	16.4%	41.0%

Unit: Millions of dollars

6.2. Greenhouse Gas Emissions in the Metropolitan Areas

6.2.1. Single-Region Model Approach

Using a single-region model, the direct and total GHG emissions for the three metropolitan areas are calculated with respect to four concepts of the environmental responsibility of a regional economy: territorial, production-based, consumption-based, and full regional environmental responsibilities. The choice of technical coefficients and the composition of the final demand term for each concept of environmental responsibility are shown in Table 11 in Chapter 4.

The per capita direct and total GHG emissions obtained from the single-region IO approach regarding the four concepts of regional environmental responsibility are displayed in Table 27. The per capita GHG emissions in the Atlanta metropolitan area are significantly higher than those in the other two metropolitan areas. For example, in a case of territorial environmental responsibility, the direct GHG emissions of the Atlanta metropolitan area are more than double those of the other areas. The major reason for these differences is the heavy reliance on fossil-fuel combustion for electricity generation in the Atlanta metropolitan area. The total GHG emissions in the utility sector in the Atlanta metropolitan area are 5.82 tons per capita in the territorial responsibility while those in the utility sector in the San Francisco and Seattle metropolitan areas are just 0.68and 0.86 ton per capita, respectively.

A comparison of the per capita GHG emissions of production-based to consumption-based regional environmental responsibilities shows that the former are higher than the latter in all three metropolitan cases, primarily because the number of exports is much higher than the number of imported products for regional consumption in all three areas, shown in Table 26.

The per capita GHG emissions of the territorial environmental responsibility occupy a relatively small proportion of the full regional environmental responsibility. The share of the total GHG emissions of territorial environmental responsibility of those of full regional environmental responsibility is 30.3% in the Atlanta metropolitan area, 23.6% in the San Francisco metropolitan area, and 26.4% in the Seattle metropolitan area. These results showed that a metropolitan economy is highly inter-connected with other regions, and the majority of GHG emissions in the metropolitan economy are related to the inter-regional flows of commodities that are either exported to meet the demand outside of a region or imported to meet demand inside of a region.

With regard to industry sectoral contributions, the utility, manufacturing, and transportation sectors are the three major sources of GHG emissions. Their combined share of the full environmental responsibility is 85.9% in the Atlanta metropolitan area, 73.6% in the San Francisco metropolitan area, and 69.7% in the Seattle metropolitan area.

Table 27: Regional GHG Emissions Responsibility from the Single-Region IO Model

Industry Sector	Territorial Environmental Responsibility		Regional Production-based Environmental Responsibility		Regional Consumption-Based Environmental Responsibility		Full Regional Environmental Responsibility	
	Direct	Total	Direct	Total	Direct	Total	Direct	Total
Atlanta Metropolitan Area								
Agriculture	0.00	0.02	0.00	0.04	0.03	0.10	0.03	0.15
Mining	0.00	0.02	0.02	0.04	0.00	0.05	0.02	0.09
Utility	4.54	5.82	8.81	11.42	4.86	7.99	9.13	15.17
Construction	0.66	0.69	0.66	0.71	0.69	0.74	0.69	0.79
Manufacturing	0.08	0.70	1.01	2.15	0.51	2.93	1.44	6.32
Wholesale/Retail	0.34	0.40	0.62	0.73	0.35	0.48	0.64	0.86
Transportation	0.34	0.68	2.38	3.04	0.62	1.42	2.66	4.17
Service	0.37	0.60	0.86	1.37	0.49	1.05	0.99	2.09
Government	0.07	0.09	0.08	0.14	0.12	0.18	0.14	0.25
Total	**6.41**	**9.03**	**14.46**	**19.63**	**7.68**	**14.93**	**15.74**	**29.89**
San Francisco Metropolitan Area								
Agriculture	0.00	0.02	0.02	0.05	0.04	0.17	0.06	0.22
Mining	0.00	0.06	0.01	0.41	0.00	0.11	0.02	0.54
Utility	0.38	0.68	0.89	1.62	0.98	1.63	1.49	2.85
Construction	0.48	0.51	0.57	0.63	0.48	0.54	0.57	0.67
Manufacturing	0.49	1.10	4.68	6.23	0.83	2.50	5.02	8.61
Wholesale/Retail	0.39	0.47	0.47	0.60	0.44	0.59	0.51	0.77
Transportation	0.44	0.70	1.60	2.21	0.62	1.22	1.79	2.99
Service	0.55	0.84	1.05	1.70	0.68	1.26	1.18	2.35
Government	0.19	0.24	0.39	0.50	0.21	0.32	0.42	0.62
Total	**2.92**	**4.62**	**9.70**	**13.95**	**4.29**	**8.34**	**11.07**	**19.62**
Seattle Metropolitan Area								
Agriculture	0.01	0.05	0.07	0.17	0.04	0.18	0.09	0.36
Mining	0.00	0.02	0.02	0.06	0.00	0.10	0.03	0.20
Utility	0.64	0.86	1.45	1.86	0.74	1.29	1.55	2.56
Construction	0.52	0.55	0.76	0.81	0.52	0.58	0.76	0.87
Manufacturing	0.15	0.72	0.98	1.98	0.64	2.49	1.47	5.01
Wholesale/Retail	0.38	0.46	0.55	0.69	0.42	0.58	0.59	0.88
Transportation	0.57	0.94	2.53	3.29	0.78	1.72	2.74	4.57
Service	0.49	0.76	0.94	1.47	0.63	1.34	1.08	2.38
Government	0.18	0.23	0.30	0.41	0.22	0.34	0.34	0.57
Total	**2.95**	**4.58**	**7.62**	**10.75**	**3.98**	**8.63**	**8.65**	**17.40**

Unit: Tons per capita

6.2.2. Two-Region Model Approach

The two-region environmental IO modeling framework quantifies the per capita GHG emissions for the three metropolitan areas. As displayed in Table 13, the drivers of the model are three types of final demand: the consumption of regionally-produced products, the consumption of imported products, and exported products. In the two-region model, the regional consumption and exports of regionally-produced products are the final demand of each metropolitan area while the consumption of imported products is the final demand of the rest of the nation. The results of a two-region IO modeling approach are categorized by the final demand component shown in Table 28.

The overall patterns of per capita GHG emissions considerably differ among three metropolitan areas. The per capita total GHG emissions of the Atlanta metropolitan area are the highest among the three metropolitan areas: 29.41 tons per capita in the Atlanta metropolitan area, 27.86 tons per capita in the San Francisco metropolitan area, and 21.55 tons per capita in the Seattle metropolitan area. As noted in the single-region approach, the heavy reliance on fossil fuel in utility sectors in the Atlanta metropolitan area is a main cause of differences.

By the final demand component, the export of regionally-produced products is the largest contributor of GHG emissions in all three metropolitan areas: 50.2% (14.74 tons per capita) of the total GHG emissions in the Atlanta metropolitan area, 47.4% (13.21 tons per capita) in the San Francisco metropolitan area, and 46.9% (10.10 tons per capita) in the Seattle metropolitan area. GHG emissions pertaining to consumption of imported commodities is also notable, but the size is relatively smaller than those of the other final demand components: 13.0% (3.82 tons per capita) of the total GHG emissions in the Atlanta metropolitan area, 25.6% (7.14 tons per capita) in the San Francisco metropolitan area, and 21.1% (4.54 tons per capita) in the Seattle metropolitan area.

Some variations occur in the geographical patterns of the GHG emissions in the three metropolitan areas. Two-thirds of the total GHGs emitted within the Atlanta metropolitan boundary (66.9% and 19.67 tons per capita). By contrast, less than half of total GHG emissions were generated within the San Francisco metropolitan boundary (45.4% and 12.63 tons per capita) and the Seattle metropolitan boundary (47.3% and 10.19 tons per capita). Despite several variations, the results of the two-regional environmental IO model indicated that the metropolitan production and consumption activities tend to exert substantial environmental pressure outside of a region. In particular, the results showed how the production activities inside of a region affect the amount of GHG emissions outside of a region. Table 28 indicates that the production activities of exported products within each metropolitan area result in considerable GHG emissions in the rest of the nation: 3.69 tons per capita (Atlanta metropolitan area), 4.61 tons per capita (San Francisco metropolitan area), and 3.99 tons per capita (Seattle metropolitan area).

Table 28 also displays sectoral contributions to GHG emissions. Similar to the single-region model, the utility, manufacturing, and transportation sectors are main sources of GHG emissions in the three areas. The combined percentages of the three sectors in total emissions are 84.9% in the Atlanta metropolitan area, 82.0% in the San Francisco metropolitan area, and 75.3% in the Seattle metropolitan area. In the cases of the San Francisco and Seattle metropolitan areas, most of the GHG emissions from the utility sector are generated in the rest of the nation. In addition, many of GHG emissions pertaining to manufacturing sectors are also generated in the rest of the nation in cases of the Atlanta (76.4%) and Seattle metropolitan (76.1%) areas. In the case of San Francisco metropolitan area, a significant portion of GHG emissions from manufacturing sectors were generated within its metropolitan boundary because of the production and the export of petroleum products.

Per capita GHG emissions are re-grouped according to the four concepts of regional environmental responsibility shown in Table 29.[28] The GHG emissions of territorial environmental responsibility are approximately a third of those of the full environmental responsibility: 36.8% (Atlanta metropolitan area), 27.0% (San Francisco metropolitan area), and 32.0% (Seattle metropolitan area). Total GHG emissions in the regional production-based environmental responsibility are consistently higher than those in the regional consumption-based environmental responsibility among the three metropolitan areas. GHG emissions regarding the four principles of environmental responsibility of a regional economy will be further discussed in the next section.

[28] GHG emissions in the territorial environmental responsibility are simply equivalent to those from the consumption of regionally-produced products, and the GHG emissions of the full regional environmental responsibility are equivalent to those in the total, the fourth column in Table 28. GHG emissions of the regional production-based environmental responsibility are the sum of the GHG emissions from consumption and export of regionally-produced products. GHG emissions in the regional consumption-based environmental responsibility are the sum of GHG emissions from the consumption of regionally-produced and imported products.

Table 28: Regional GHG Emissions from the Two-Region IO Model

	Final Demand											
	Consumption of Regionally-Produced Products			Export of Regionally Produced Products			Consumption of Imported Products			Total		
	Metro	**RN***	**Sum**	**Metro**	**RN**	**Sum**	**Metro**	**RN**	**Sum**	**Metro**	**RN**	**Sum**
Atlanta Metropolitan Area												
Agriculture	0.01	0.06	0.07	0.01	0.07	0.07	0.00	0.10	0.10	0.01	0.23	0.24
Mining	0.00	0.05	0.05	0.02	0.06	0.08	0.00	0.03	0.03	0.02	0.14	0.16
Utility	5.98	0.67	6.65	6.30	1.00	7.30	0.01	1.32	1.33	12.30	2.98	15.28
Construction	0.69	0.01	0.70	0.03	0.02	0.05	0.00	0.04	0.04	0.72	0.08	0.80
Manufacturing	0.19	1.02	1.21	1.09	1.88	2.97	0.00	1.27	1.27	1.28	4.16	5.45
Wholesale/Retail	0.41	0.02	0.43	0.36	0.03	0.39	0.00	0.07	0.07	0.77	0.12	0.89
Transportation	0.63	0.25	0.88	2.45	0.35	2.80	0.01	0.57	0.57	3.09	1.17	4.26
Service	0.57	0.15	0.72	0.78	0.23	1.01	0.00	0.30	0.30	1.35	0.68	2.03
Government	0.09	0.03	0.12	0.04	0.05	0.09	0.00	0.10	0.10	0.13	0.18	0.31
Industry Sum	**8.56**	**2.26**	**10.83**	**11.08**	**3.69**	**14.76**	**0.03**	**3.80**	**3.82**	**19.67**	**9.74**	**29.41**
San Francisco Metropolitan Area												
Agriculture	0.01	0.07	0.07	0.02	0.05	0.06	0.00	0.14	0.14	0.02	0.25	0.27
Mining	0.00	0.04	0.05	0.03	0.04	0.07	0.00	0.03	0.03	0.03	0.12	0.15
Utility	0.52	1.82	2.34	0.71	2.38	3.09	0.00	4.50	4.50	1.23	8.71	9.93
Construction	0.51	0.01	0.53	0.13	0.02	0.15	0.00	0.02	0.02	0.64	0.06	0.70
Manufacturing	0.81	0.99	1.80	5.02	1.33	6.35	0.01	1.26	1.28	5.85	3.58	9.43
Wholesale/Retail	0.48	0.03	0.51	0.16	0.03	0.20	0.00	0.12	0.12	0.64	0.18	0.82
Transportation	0.63	0.34	0.96	1.42	0.48	1.91	0.00	0.62	0.62	2.05	1.44	3.49
Service	0.81	0.18	0.99	0.84	0.24	1.07	0.00	0.37	0.37	1.65	0.79	2.44
Government	0.24	0.02	0.27	0.28	0.03	0.31	0.00	0.06	0.06	0.52	0.11	0.63
Industry Sum	**4.01**	**3.51**	**7.52**	**8.60**	**4.61**	**13.21**	**0.02**	**7.11**	**7.14**	**12.63**	**15.23**	**27.86**
Seattle Metropolitan Area												
Agriculture	0.04	0.06	0.10	0.13	0.06	0.19	0.00	0.12	0.12	0.17	0.24	0.41
Mining	0.00	0.06	0.06	0.03	0.07	0.10	0.00	0.03	0.03	0.03	0.17	0.19
Utility	0.88	0.95	1.83	1.09	1.33	2.42	0.00	1.90	1.90	1.97	4.18	6.15
Construction	0.56	0.01	0.57	0.29	0.02	0.30	0.00	0.02	0.02	0.84	0.06	0.90
Manufacturing	0.30	1.12	1.42	1.00	1.65	2.65	0.00	1.37	1.38	1.30	4.14	5.44
Wholesale/Retail	0.46	0.03	0.50	0.26	0.05	0.31	0.00	0.10	0.11	0.73	0.18	0.91
Transportation	0.89	0.33	1.22	2.42	0.45	2.87	0.00	0.56	0.56	3.31	1.33	4.65
Service	0.70	0.24	0.94	0.71	0.32	1.03	0.00	0.38	0.38	1.41	0.93	2.34
Government	0.24	0.03	0.27	0.19	0.04	0.23	0.00	0.06	0.06	0.43	0.13	0.56
Industry Sum	**4.07**	**2.83**	**6.90**	**6.11**	**3.99**	**10.10**	**0.01**	**4.54**	**4.55**	**10.19**	**11.36**	**21.55**

* RN is an abbreviation for the "rest of the nation."
Unit: Tons per capita

Table 29: Regional GHG Emissions of the Four Concepts of Regional Environmental Responsibility from the Two-Region IO Model

	Territorial Environmental Responsibility			Regional Production-based Environmental Responsibility			Regional Consumption-based Environmental Responsibility			Full Regional Environmental Responsibility		
	Metro	**RN***	**Sum**	**Metro**	**RN**	**Sum**	**Metro**	**RN**	**Sum**	**Metro**	**RN**	**Sum**
Atlanta Metropolitan Area												
Agriculture	0.01	0.06	0.07	0.01	0.13	0.14	0.01	0.16	0.17	0.01	0.23	0.24
Mining	0.00	0.05	0.05	0.02	0.11	0.13	0.00	0.08	0.08	0.02	0.14	0.16
Utility	5.98	0.67	6.65	12.28	1.67	13.95	5.99	1.98	7.98	12.30	2.98	15.28
Construction	0.69	0.01	0.70	0.72	0.03	0.75	0.69	0.06	0.74	0.72	0.08	0.80
Manufacturing	0.19	1.02	1.21	1.28	2.90	4.18	0.20	2.28	2.48	1.28	4.16	5.45
Wholesale/Retail	0.41	0.02	0.43	0.77	0.05	0.82	0.41	0.09	0.50	0.77	0.12	0.89
Transportation	0.63	0.25	0.88	3.08	0.60	3.68	0.64	0.82	1.46	3.09	1.17	4.26
Service	0.57	0.15	0.72	1.35	0.38	1.73	0.57	0.45	1.02	1.35	0.68	2.03
Government	0.09	0.03	0.12	0.13	0.08	0.21	0.09	0.13	0.22	0.13	0.18	0.31
Industry Sum	**8.56**	**2.26**	**10.83**	**19.64**	**5.95**	**25.59**	**8.59**	**6.06**	**14.65**	**19.67**	**9.74**	**29.41**
San Francisco Metropolitan Area												
Agriculture	0.01	0.07	0.07	0.02	0.11	0.14	0.01	0.20	0.21	0.02	0.25	0.27
Mining	0.00	0.04	0.05	0.03	0.09	0.12	0.00	0.08	0.08	0.03	0.12	0.15
Utility	0.52	1.82	2.34	1.23	4.20	5.43	0.52	6.32	6.84	1.23	8.71	9.93
Construction	0.51	0.01	0.53	0.64	0.03	0.67	0.52	0.04	0.55	0.64	0.06	0.70
Manufacturing	0.81	0.99	1.80	5.83	2.32	8.15	0.83	2.25	3.08	5.85	3.58	9.43
Wholesale/Retail	0.48	0.03	0.51	0.64	0.06	0.70	0.48	0.15	0.63	0.64	0.18	0.82
Transportation	0.63	0.34	0.96	2.05	0.82	2.87	0.63	0.95	1.58	2.05	1.44	3.49
Service	0.81	0.18	0.99	1.65	0.42	2.07	0.81	0.55	1.36	1.65	0.79	2.44
Government	0.24	0.02	0.27	0.52	0.06	0.57	0.24	0.08	0.32	0.52	0.11	0.63
Industry Sum	**4.01**	**3.51**	**7.52**	**12.61**	**8.11**	**20.72**	**4.04**	**10.62**	**14.66**	**12.63**	**15.23**	**27.86**
Seattle Metropolitan Area												
Agriculture	0.04	0.06	0.10	0.17	0.12	0.29	0.04	0.18	0.22	0.17	0.24	0.41
Mining	0.00	0.06	0.06	0.03	0.13	0.16	0.00	0.09	0.09	0.03	0.17	0.19
Utility	0.88	0.95	1.83	1.97	2.28	4.25	0.88	2.85	3.73	1.97	4.18	6.15
Construction	0.56	0.01	0.57	0.84	0.03	0.88	0.56	0.04	0.59	0.84	0.06	0.90
Manufacturing	0.30	1.12	1.42	1.30	2.77	4.07	0.30	2.49	2.80	1.30	4.14	5.44
Wholesale/Retail	0.46	0.03	0.50	0.73	0.08	0.81	0.46	0.14	0.60	0.73	0.18	0.91
Transportation	0.89	0.33	1.22	3.31	0.78	4.09	0.90	0.88	1.78	3.31	1.33	4.65
Service	0.70	0.24	0.94	1.41	0.55	1.96	0.70	0.61	1.31	1.41	0.93	2.34
Government	0.24	0.03	0.27	0.43	0.07	0.50	0.24	0.09	0.33	0.43	0.13	0.56
Industry Sum	**4.07**	**2.83**	**6.90**	**10.18**	**6.82**	**17.00**	**4.08**	**7.37**	**11.46**	**10.19**	**11.36**	**21.55**

* RN is an abbreviation for the “rest of the nation.”
Unit: Tons per capita

6.2.3. Comparison of the Single- and Two-region Input-Output Approaches

This section compares the results of the single- and two-region IO modeling frameworks. The two-regional approach is expected to more accurately assess the amount of GHG emissions than the single-region approach because it takes into account the inter-regional effect and uses regional-specific GHG emission coefficients. The GHG emissions calculated from the two modeling approaches are compared in Figure 16.

Overall, the comparison shows a considerable difference between the estimates of GHG emissions of the two approaches. With respect to territorial environmental responsibility and regional production-based environmental responsibility, the GHG emissions of the two-region approach are consistently higher than those of the single-region approach. The consideration of the effect of inter-regional linkages in the two-region approach is a primary cause of those differences. Since the single-region model employed intra-region input coefficients in the territorial-and production-based responsibilities, the single-region model does not account for GHG emissions originating in the consumption of intermediate input products supplied inter-regionally.

In consumption-based and full environmental responsibilities, the single-region model utilized regional technical coefficients. Consequently, the difference of GHG emissions calculated by the single- and two-region approaches is small in the Atlanta metropolitan area where the metro-specific GHG emission coefficients of major energy-intensive sectors are similar to national GHG emission coefficients. However, in the San Francisco and Seattle areas, the GHG emissions calculated by the single-regional approach are still lower than those calculated by the two-region approach. The main cause of differences is the lower GHG emission coefficients of the major energy-intensive sectors in the San Francisco and Seattle metropolitan area. The extent to which the estimation errors of the single-region model will increase in cases in which the metropolitan-specific energy use coefficients abnormally diverge from the national average and the share of consumption of imported products is relatively large.

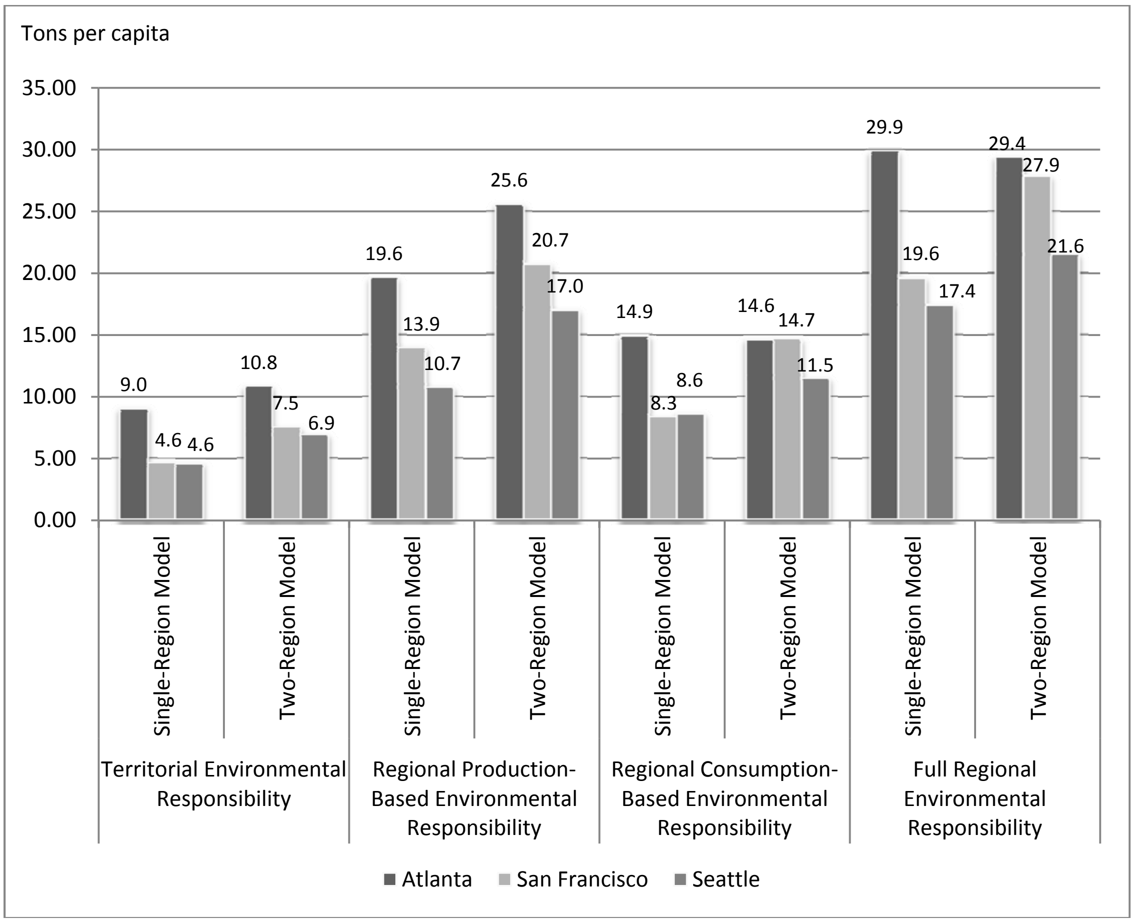

Figure 16: Comparison of the GHG Emissions Calculated by the Single- and Two-Region IO Models

The results of the regional environmental IO modeling frameworks provide several insights into the regional GHG emission inventory study. For one, the results indicated that per capita GHG emissions considerably differ from region to region.

Region-specific energy use and GHG emission coefficients are essential to a regional GHG emission inventory study. Second, the results also showed that exported and imported products play a significant role in the level of metropolitan GHG emissions. Approximately two-thirds of GHG emissions in the full regional environmental responsibility are directly or indirectly associated with exported or imported products in all three metropolitan area cases. The two-region approach showed that a significant portion (one-third or more than half) of GHG emissions in the full regional environmental responsibility is actually generated outside of their geographical metropolitan boundary. These findings emphasize the significance of quantifying cross-boundary GHG emissions in the regional GHG emission inventory study. Finally, the results showed the considerable differences in estimates of regional GHG emissions between single- and two-region approaches. Specifically, when the energy use pattern of the studied region largely diverges from the national average, the estimation errors will be significant in the single-region environmental IO model. Thus, those results accentuate the necessity of the development of a multi-region environmental IO model.

CHAPTER 7. SIMULATION OF THE ECONOMIC AND ENVIRONMENTAL IMPACT

The intention of this chapter is to estimate the economic and environmental impact of recycling industrial activities through simulations of the two-region environmental IO model. To evaluate the net effect, the simulations consider the effect of not only recycling industrial activities but also industrial activities displaced by recycling. To carry out the simulations, this chapter first establishes the economic and environmental profile of recycling technology by compiling data from governmental statistics, prior studies of process-based engineering models, survey on the waste carpet recycling[29], and personal communication with recycling businesses and experts. It also constructs transportation scenarios fitted to each case of the geographical area that a recycling facility serves. Finally, it presents the net economic and environmental impact of four cases: mixed CDW recycling, deconstruction, recycled nylon 6 production, and recycled carpet padding production.

7.1. Construction and Demolition Waste Recycling

This section analyzes the economic and environmental impact of mixed CDW recycling. The San Francisco metropolitan area is a case study area with well-defined institutional regulations and a well-developed recycling infrastructure. The focus of this section is two-fold. First, it examines a mixed CDW recycling facility through which most CDW is processed in the San Francisco metropolitan area. The mixed CDW

[29] The survey method and results are summarized in the Appendix.

recycling facility is expected to provide additional job opportunities and to reduce energy use by replacing virgin materials. Second, it also investigates the impact of deconstruction. Although the deconstruction technique has limited application in practice, it is recognized as an alternative sustainable CDW management approach that requires more labor input and less energy use compared to the traditional demolition technique. The simulation computes the economic and environmental impact when the deconstruction technique applies to residential building dismantling in the San Francisco metropolitan area.

7.1.1. Mixed Construction and Demolition Waste Recycling in the San Francisco Metropolitan Area

Scenario of Mixed CDW Recycling and Disposal Paths

This simulation compares recycling and disposal paths pertaining to the management of mixed CDW as illustrated in Figure 17. The disposal path, consisting of industrial activities that could be replaced by recycling activities, ought to be considered when the net economic and environmental impact of mixed CDW recycling is assessed. The economic and environmental impact of the disposal path will be subtracted from the impact of the recycling path for estimating the net effect.

The recycling path involves three major industrial activities. First, a waste hauling company transports diverted CDW using roll-off containers from construction or demolition sites to a recycling facility and also transports non-recyclable components from a mixed CDW recycling facility to landfills. The next industrial activity is a mixed CDW processing facility that plays a central role in the sustainable management of

CDW because a great deal of CDW is discarded without source separation. The mixed CDW recycling facility sorts the CDW by size and type through a screen, a magnetic separator, and hand-picking processes. The sorted materials typically include ferrous metal, dirt and sand, bulky concrete, paper and cardboard, and lumber and diverse wood-related products. The bulky asphalt and concrete are crushed into smaller pieces and the clean wood-related products are ground into wood chips. Reusable lumber is salvaged. Both salvaged and recycled materials are sold back to construction companies or consumers or utilized as feedstock in other industrial processes. Finally, the non-recyclable components are accepted by landfill operators and potentially used as alternative daily cover or disposed in landfills. The simulation first calculates the economic and environmental consequences of these three industrial activities in the recycling path.

The simulation also considers the impact of industrial activities displaced by the recycling industrial activities in the disposal path, which consists of hauling waste to landfills, disposing the CDW in landfills, and producing virgin material. The displaced virgin materials investigated here comprise natural aggregate (i.e., IMPLAN 25, sand, gravel, clay, and refractory mining) and wood products (i.e., IMPLAN 112, sawmills).

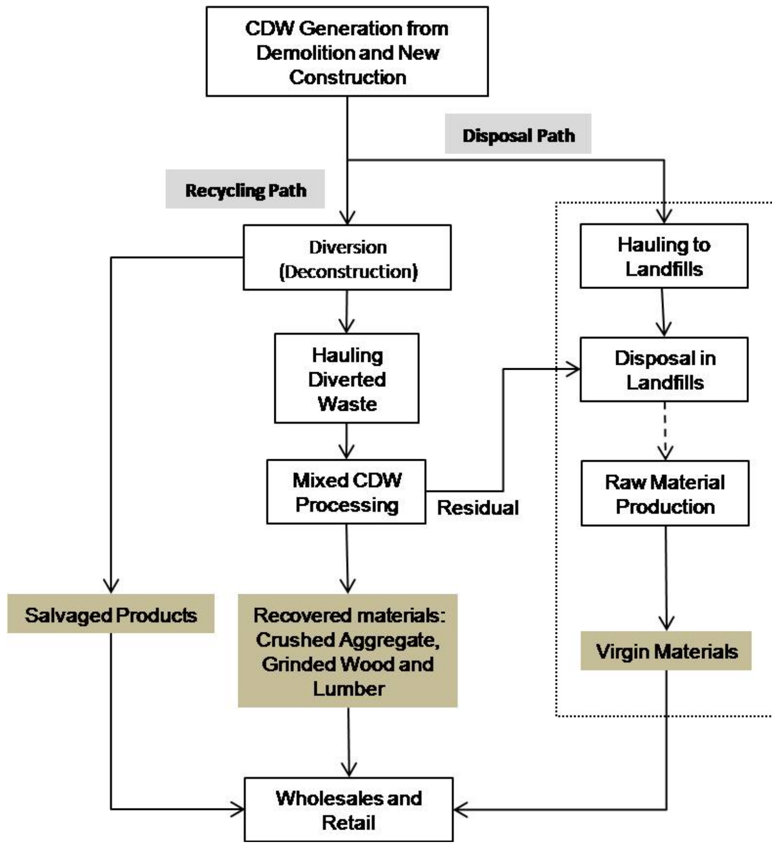

Figure 17: Recycling and Disposal Path of Mixed Construction and Demolition Waste

Process of Mixed CDW Recycling

Since no industry sector in the existing IO classification represents the industrial activity of a mixed CDW recycling facility, a new industry sector, mixed CDW recycling, needs to be added in the existing regional IO framework. Hence, this study develops a typical economic and environmental profile of a CDW recycling facility, which requires data pertaining to the economic input structure, employment, wages, energy use, and greenhouse gas emissions. These data are obtained from various sources: the economic input structure from a local government report (North Central Texas Council of Governments, 2007), the volume and the composition of processed

CDW from personal communication with the staff of the Alameda County Waste Management Authority and waste characterization study (City of San Jose, 2008), transporting and operating costs from the reports of the South Bayside Waster Management Authority (SBWMA), and the energy use of CDW recycling from a prior LCA study (Levis, 2008). As shown in Figure 23, energy consumption, labor input, and GHG emissions are normalized by millions of dollars in revenue in order to fit into environmental IO modeling.

A prototype mixed CDW recycling facility can process 85,000 tons of CDW per year and produce recycled aggregate, salvaged lumber, and ground wood chips potentially used for mulch and bio fuel. Although ferrous metal and baled paper are also recovered materials, this analysis does not consider the impact of recovered metal and paper because they consist of a small portion, only 2 to 4 percent by weight. The facility employs 3.95 workers per million dollars of revenue. The positions consist of site supervisors, equipment operators, mechanics, manual picking line workers, sales personnel, and administrators. Diesel fuel is a main energy source for rolling stock such as loaders, excavators, and forklifts, and the electricity is a primary energy source for processing equipment such as conveyers, magnets, and trommels. The operation of this facility consumes 2.655 TJ per million dollars of revenue of petroleum-based fuel and 0.3555 Mkwh per million dollars of revenue. The CDW recycling facility emits 187 tons of GHGs per million dollars of revenue from the direct combustion of diesel fuel.

Figure 18: Input and Output Flow of the Mixed CDW Recycling Facility

Mixed CDW Collection System

The mixed CDW recycling facility serves the metropolitan area. To compare the impact of the CDW collection system in the recycling and disposal paths, this research estimated the average shipping distance, the total travel distance, and associated energy consumption and GHG emissions in a case of the San Francisco metropolitan area. The physical location of CDW recycling facilities, transfer stations, and landfills are mapped in Figure 4. For simplicity of the CDW collection system, this research makes one assumption. That is, the average weight of CDW produced per capita typically remains constant across a metropolitan area. As shown in Table 1, the San Francisco metropolitan area generates 0.203 tons/person/year of CDW on average. The census tract is a spatial unit of the origin of CDW generation for transportation. The weight of CDW generated in each census tract is proportional to the population of a census tract.

The recycling path consists of two types of trips. The CDW generated in each census tract is shipped to the nearest recycling facility, and the residual of a recycling facility is shipped to the nearest landfills. By contrast, in a disposal path, the CDW generated in each census tract is shipped to the nearest transfer stations or landfills. If it enters transfer stations, it also travels to the nearest landfills for disposal.

Based on the location information in Figure 4 and the highway planning network map provided by the U.S Department of Transportation,[30] four types of road distances are calculated using the network analyst tool in ArcMap software that allows us to create an origin-destination matrix: 1) the distance from the geometric center of a census tract to the nearest recycling facility, 2) the distance from a recycling facility to the nearest landfills, 3) the distance from the geometric center of a census tract to the nearest landfills or transfer station, and 4) the distance from a transfer station to the nearest landfills. Since many CDW recycling facilities co-locate with transfer stations in the San Francisco metropolitan area, the average shipping distances between the recycling and disposal paths are similar. The average road distance of a recycling path is 5.8 miles from a census tract to the nearest recycling facility and 15.0 miles from a recycling facility to the nearest landfills whereas the average road distance of a disposal path is 4.8 miles from census tract to the nearest transfer station or landfills and 17.4 miles from transfer station to the nearest landfills.

The average shipping distance, the vehicle type, the bin size, and number of trips are primary determinants of energy use in CDW transportation.[31] In this analysis, a roll-off bin with 10 tons and heavy duty diesel vehicle (HDDV) 8A are assumed to be used to transport CDW. The fuel economy of HDDV8A is 6.8 mpg (Guidry, 2008). Using the information about the bin size, the CDW weight, and average shipping distances, this research calculates the total travel distance required to transport the CDW to particular

[30] The highway network map is downloaded from a website of the U.S. Department of Transportation, accessed April 4, 2012 at http://www.fhwa.dot.gov/planning/nhpn/

[31] The effects of bin size and type of vehicle on energy use in transportation were explored in the case of waste carpet (Guidry, 2008).

destinations.[32] Energy consumption is calculated by multiplying the total travel distance by the fuel economy of the transporting vehicles, and GHG emissions are also calculated by multiplying energy consumption by the GHG emission factor of distillate fuel oil (U.S. EPA, 2011). The average shipping distance, the total travel distance, energy use, and GHG emissions are summarized in Table 30. The total travel distance of the recycling path is far lower than that of the disposal path by 118,894 miles. Consequently, the total energy consumption and GHG emissions in a recycling path are less than half those in the disposal path.

Table 30: Comparison of CDW Transportation Scenarios of the Recycling and Disposal Paths

	CDW Shipped (Tons)	Average Distance (Miles)	Total Travel Distance (Miles)	Total Energy Use (TJ)	Total GHG Emissions (Tons)
Recycling Path					
Census Tract - Recycling Facility	85,000	5.8	62,464	1.3	94
Recycling Facility – Landfills	21,250	15.0	44,276	1.0	67
Total	106,250		106,740	2.3	161
Disposal Path					
Census Tract – Transfer Station/Landfills	85,000	4.8	51,829	1.1	78
Transfer Station - Landfills	78,830	17.4	173,805	3.7	262
Total	163,830		225,634	4.9	340

Simulation Results

The simulation estimates the economic and environmental impact of industrial activities in the recycling and disposal paths. The results with the aggregate industry

[32] The total travel distance is calculated by multiplying the required number of trips by average shipping distance. The required numbers of trips is obtained by dividing the total CDW by the size of the bin.

sectors are summarized in Table 31. The simulation shows the net increases in jobs and incomes as well as the net reduction in energy use and GHG emissions. The mixed CDW recycling facility directly hires 35 employees. The total economy-wide employment impact is 73 in the recycling path whereas that of the disposal path is 67. The net effect of employment is 7. The effect of job creation in the CDW recycling path is offset by sectors whose industrial activities are displaced: primary (-9), sawmill (-5), CDW collection (-6), landfills (-16), and service (-25). The net income effect is $0.318 million. The net outcome impact is negative, -$1.724 million, primarily because of the price difference between virgin and recycled materials.

With respect to the environmental impact, the magnitude of energy savings from mixed CDW recycling is relatively small. The total net energy savings is 15 TJ when 85,000 tons of CDW is diverted and recycled. The primary (logging), sawmill, and landfill sectors are the main contributors of energy savings. Because displaced industry sectors such as sand and gravel and sawmills are not energy-intensive and the energy savings from the transportation of CDW is relatively small, the effect of energy savings is not significant. The reduction of the total GHG emissions from 85,000 tons of CDW recycling is 1,385 tons.

Table 31: Economic and Environmental Impact of the Mixed CDW Recycling Facility

	San Francisco Metropolitan Area					Rest of the Nation					Total				
	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)
Recycling Path															
Primary	0.023	0	0.002	0	1	0.249	1	0.036	2	168	0.272	1	0.038	3	170
Sand and Gravel	0.000	0	0.000	0	0	0.001	0	0.000	0	0	0.001	0	0.000	0	0
Utility	0.080	0	0.012	1	65	0.033	0	0.005	2	154	0.113	0	0.017	3	219
Construction	0.021	0	0.010	0	3	0.013	0	0.005	0	2	0.034	0	0.015	0	6
Manufacturing	1.067	1	0.063	6	368	1.158	2	0.163	5	281	2.225	3	0.226	10	648
Sawmills	0.012	0	0.001	0	4	0.138	1	0.021	1	35	0.151	1	0.022	1	38
CDW Collection	0.695	2	0.137	2	165	0.059	0	0.017	0	30	0.754	3	0.154	3	195
Landfills	3.080	5	0.355	8	563	0.253	1	0.072	2	136	3.333	6	0.427	10	698
Service	2.196	14	0.717	8	396	1.438	10	0.449	2	153	3.634	24	1.166	10	549
Mixed CDW Recycling	8.860	35	1.693	30	1,657	0.000	0	0.000	0	0	8.860	35	1.693	30	1,657
Total	16.033	57	2.992	55	3,221	3.342	17	0.767	14	959	19.376	73	3.759	70	4,180
Disposal Path															
Primary	0.102	0	0.007	0	20	1.066	8	0.174	18	1,236	1.168	9	0.181	19	1,283
Sand and Gravel	0.287	1	0.112	1	75	0.007	0	0.002	0	3	0.294	1	0.114	1	78
Utility	0.118	0	0.020	1	52	0.034	0	0.005	2	142	0.152	0	0.025	3	194
Construction	0.023	0	0.010	0	4	0.014	0	0.005	0	3	0.037	0	0.015	0	6
Manufacturing	1.728	2	0.115	9	558	0.918	2	0.109	4	233	2.646	4	0.224	12	791
Sawmills	1.471	4	0.120	8	449	0.159	1	0.024	1	40	1.630	5	0.144	8	489
CDW Collection	1.570	5	0.309	5	372	0.113	1	0.033	1	58	1.683	6	0.342	6	430
Landfills	9.469	15	1.092	25	1,730	0.204	1	0.058	2	109	9.673	16	1.150	26	1,839
Service	2.269	14	0.766	6	289	1.508	11	0.470	3	159	3.776	25	1.236	8	448
Mixed CDW Recycling	0.041	0	0.008	0	8	0.000	0	0.000	0	0	0.041	0	0.008	0	8
Total	17.078	43	2.560	55	3,556	4.022	23	0.880	30	2,009	21.100	67	3.440	84	5,565

Table 31 continued

	San Francisco Metropolitan Area					Rest of the Nation					Total				
	Output	Employment	Income	Energy	GHG	Output	Employment	Income	Energy	GHG	Output	Employment	Income	Energy	GHG
	($M)	(Persons)	($M)	(TJ)	(Tons)	($M)	(Persons)	($M)	(TJ)	(Tons)	($M)	(Persons)	($M)	(TJ)	(Tons)
Net Effect															
Primary	-0.079	0	-0.005	0	-19	-0.817	-7	-0.138	-16	-1,095	-0.897	-7	-0.143	-16	-1,113
Sand and Gravel	-0.287	-1	-0.112	-1	-75	-0.007	0	-0.002	0	-3	-0.293	-1	-0.114	-1	-77
Utility	-0.038	0	-0.008	0	13	-0.001	0	0.000	0	13	-0.039	0	-0.008	0	26
Construction	-0.002	0	0.000	0	0	-0.001	0	0.000	0	0	-0.003	0	0.000	0	0
Manufacturing	-0.661	-2	-0.052	-3	-191	0.240	1	0.054	1	48	-0.421	-1	0.002	-2	-143
Sawmills	-1.459	-4	-0.119	-8	-445	-0.020	0	-0.003	0	-5	-1.479	-4	-0.122	-8	-450
CDW Collection	-0.876	-3	-0.172	-3	-208	-0.054	0	-0.016	0	-28	-0.930	-3	-0.188	-3	-235
Landfills	-6.389	-10	-0.737	-17	-1,167	0.049	0	0.014	0	26	-6.340	-10	-0.723	-16	-1,141
Service	-0.072	-1	-0.049	2	107	-0.070	0	-0.021	0	-6	-0.142	-1	-0.070	2	101
Mixed CDW Recycling	8.819	35	1.686	30	1,649	0.000	0	0.000	0	0	8.819	35	1.686	30	1,649
Total	-1.045	14	0.432	1	-335	-0.680	-7	-0.113	-15	-1,049	-1.724	7	0.318	-15	-1,385

Sensitivity Analysis

The sensitivity analysis is conducted with a change in the substitution rate and the amount of CDW processed by mixed CDW recycling facilities. First, the baseline simulation initially assumed that the recycled material from mixed CDW recycling replaced virgin material production at a 100% substitution rate. If the substitution rate decreased, implying that recycling creates new demand, the net economic effect would increase. The analysis investigates how the economic and environmental impact will change when the substitution rate decreases to 80%, 50%, and 20%. The results of the sensitivity analysis in Table 32 show that the net job impact increases to 31 and the net income impact to $1.261 million when the substation rate declines to 20%; the positive environmental impact, however, disappears with the reduction in the substitution rate. An approximate 10% decrease in the substitution rate results in a reduction of 4TJ in energy savings. When the substitution rate is 20% (ALT3), the mixed CDW recycling consumes additional 17 TJ of energy.

The sensitivity analysis also examines the change in the economic and environmental impact when the processed amount of CDW increases. Although the total quantity of discarded CDW is uncertain because it fluctuates along with economic cycles, if 0.203 tons/yr/person apply in Table 1, the total discarded CDW per year is roughly 875,000 tons in the San Francisco metropolitan area. As a prototype CDW recycling facility in the initial simulation has a processing capacity of 85,000 tons per year, this facility is likely to process less than 10% of the total discarded CDW. Because most local ordinances require a diversion rate of 50% or more at construction and demolition sites, roughly 50 to 75 percent of CDW generated at construction and demolition sites is likely to be processed at mixed CDW recycling facilities in the San Francisco metropolitan area.

The sensitivity analysis, thus, investigates the change in the economic and environmental impact when 20% (175,194 tons in ALT 4), 50% (437,980 tons in ALT 5), and 75% (656,965 tons in ALT 6) of total discarded CDW is shipped and processed by mixed CDW recycling facilities. Total employment in a recycling path increases to 378 with a 50% diversion rate (ALT5) and 567 with a 75% diversion rate (ALT6). The total net effect of employment is 36 with a 50% diversion rate (ALT5) and 53 with a 75% diversion rate (ALT6). The net energy savings and the net GHG emission reductions are 113 TJ and 10,701 tons, respectively, with a 75% diversion rate.

Table 32: Results of the Sensitivity Analysis of Mixed CDW Recycling

		Substitution Rate			Quantity of CDW Processed		
	Initial	**ALT1**	**ALT2**	**ALT3**	**ALT4**	**ALT5**	**ALT6**
Substitution Rate	100%	80%	50%	20%	-	-	-
Quantity of CDW(Tons)	85,000	-	-	-	175,194	437,980	656,965
Recycling Path							
Output ($ Millions)	19.376	19.917	20.544	21.171	39.935	99.836	149.753
Employment (Persons)	73	76	78	81	151	378	567
Income ($ Millions)	3.759	3.869	3.996	4.123	7.747	19.367	29.051
Energy (TJ)	70	72	74	76	144	359	539
GHG (Tons)	4,180	4,298	4,435	4,571	8,616	21,539	32,308
Disposal Path							
Output ($ Millions)	21.100	20.276	19.041	17.805	43.489	108.721	163.081
Employment (Persons)	67	62	56	50	137	343	514
Income ($ Millions)	3.440	3.296	3.079	2.862	7.091	17.727	26.590
Energy (TJ)	84	78	69	59	174	434	651
GHG (Tons)	5,565	5,160	4,552	3,945	11,469	28,673	43,009
Net Effect							
Output ($ Millions)	-1.724	-0.359	1.504	3.366	-3.554	-8.885	-13.327
Employment (Persons)	7	13	22	31	14	36	53
Income ($ Millions)	0.318	0.573	0.917	1.261	0.656	1.641	2.461
Energy (TJ)	-15	-6	5	17	-30	-75	-113
GHG (Tons)	-1,385	-862	-118	626	-2,854	-7,134	-10,701

7.1.2. Effect of Deconstruction

Scenario of Deconstruction

The simulation in the previous section examined the transport, recycling, and disposal stages, but it omitted the building demolition stage. The economic and environmental consequences of CDW recycling change when different building dismantling techniques apply. As seen in the literature review chapter, even though the deconstruction technique has economic and environmental potential, the application of the deconstruction technique is limited mainly because of the time constraints of construction projects. In the San Francisco metropolitan area, a small number of for-profit companies or non-profit organizations provide deconstruction services. This simulation compares the economic and environmental impacts of the deconstruction of residential buildings to those of traditional demolition.

When the deconstruction technique applies, it has multiple effects on the CDW recycling path as displayed in Figure 19. For one, the deconstruction technique is a more labor- and less energy-intensive process than the traditional demolition technique. Hence, the impact on employment is expected to be higher, but the impact of energy use and GHG emissions of deconstruction is expected to be lower. In addition, since deconstruction diverts the greater amount of CDW, the transportation impact is expected to change. Deconstruction reduces the demand of transportation as the amount of waste sent to CDW recycling facilities and landfills decreases, but it creates new transportation demand, that is the salvaged products are transported from deconstruction sites to wholesale and retail shops. Hence, the impact on transportation is determined by the average distance of each trip and the associated weight of CDW traveled. Finally, deconstruction affects the activities of mixed CDW recycling facilities and landfills. These two industrial activities decrease because the amount of incoming CDW decreases. In summary, this simulation investigates the impact of deconstruction itself as well as its subsequent impact on transportation, mixed CDW recycling, and landfills, compared to the impact of traditional demolition.

The San Francisco Housing Inventory (San Francisco Planning Department, 2011) informs the numbers of housing units authorized, newly constructed, and demolished every year. In 2010, 170 housing units were demolished and 1,203 units authorized in the City and the County of San Francisco. By extrapolation,[33] the demolished housing units are roughly over 730 in the San Francisco metropolitan area in 2010. According to ReUse People (Reiff, 2010), one deconstruction crew consisting of five workers can dismantle 25 residential buildings per year. Initially, this research assumed that 250 units of residential buildings, approximately 33% of the total number demolished, were dismantled through deconstruction.

[33] The information about demolished housing units in the San Francisco metropolitan area has not been compiled. The demolished units and authorized units are known in the City and County of San Francisco while only authorized units are known for the metropolitan area. Thus, the demolished units in the metropolitan area are extrapolated using the ratio of authorized units and demolished units in the City and County of San Francisco.

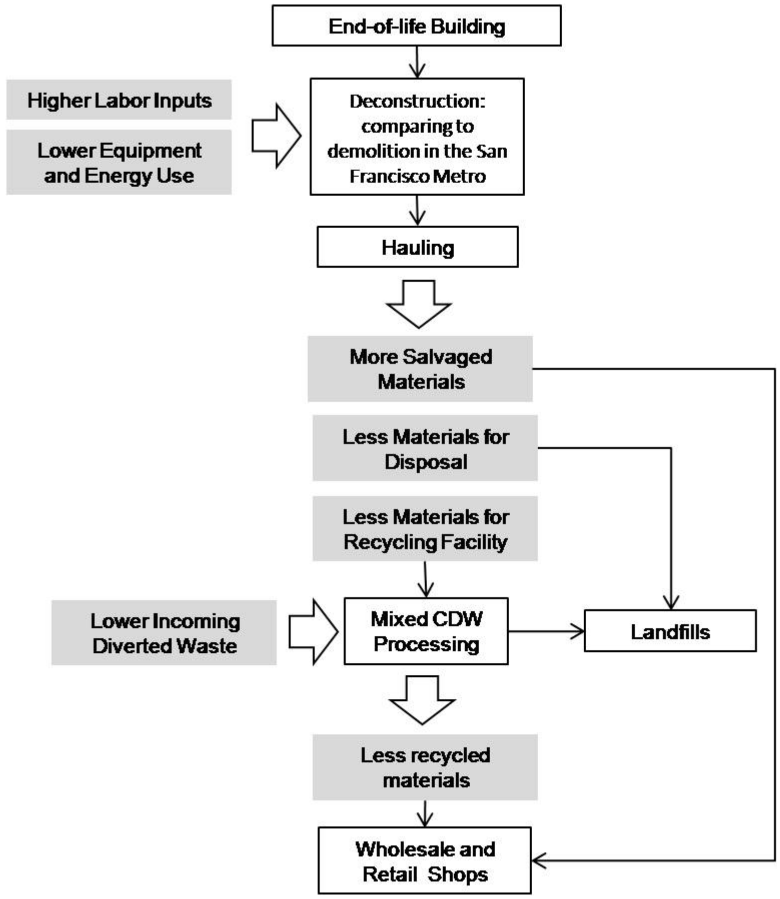

Figure 19: Effect of Deconstruction in the Flow of CDW Recycling

Comparison of Deconstruction and Demolition

To create two new sectors of demolition and deconstruction in the regional environmental IO model, this research builds an economic and environmental profile of both techniques. First, the economic and energy data about demolition are available in the Economic Census 1997and 2007 on the state and national levels, from which employment, income, energy use, and GHG emission coefficients were estimated for a demolition sector. Next, for building a comparable profile of deconstruction, the research compiled data from various sources: the salvage and diversion rates of the deconstruction of residential buildings from ReUse People (Reiff , Personal communication, 2011); the labor input and wage rates of deconstruction from a presentation by ReUse People (Reiff, 2010); the ratio of the labor input of demolition to that of deconstruction from the report of the Center for Construction and Environment at the University of Florida (Guy, 2001).

According to the estimate of ReUse People for residential deconstruction projects, the deconstruction project of a single residential building with 2,000 sq. ft. generated approximately 80 tons of CDW on average. Among the CDW, fixtures (8%) and lumber (13%) was salvaged for reuse, concrete (49%) and other material (20%) was diverted for recycling, and rest CDW (10%) was disposed.[34] Almost 90% of CDW is potentially salvaged and diverted from deconstruction. When 50% or more of the diversion requirements of local ordinances in the San Francisco metropolitan area is taken into account, roughly 15 to 40% of additional materials are salvaged or recycled in deconstruction. In this analysis, the diversion rate of demolition is assumed to be 75%, so an additional 15% of CDW is diverted from deconstruction.[35]

One important difference of demolition and deconstruction is their labor input requirements. In an investigation of six residential deconstruction projects in Florida, Guy (2001) measured the labor costs per sq. ft. for each project. According to his estimates, the labor cost ratio of deconstruction to demolition was an average of 2.09, indicating that deconstruction labor costs are more than double those of demolition. Using this ratio and employment and wage information from ReUse People, the employment and income coefficients for a deconstruction sector are estimated. Since this

[34] The average salvage and diversion rate is calculated based on nine residential deconstruction projects in the San Francisco and Seattle metropolitan areas (Reiff, 2011).
[35] As shown in Table 4, the low required diversion rate is 50% of CDW generated in a case of the City of Alameda, and the high required diversion rate is 100% of concrete and 65% of remaining waste generated in the case of the City of Oakland. Hence, 15% of additional diversion from deconstruction is a conservative rate.

ratio was calculated from six cases of deconstruction projects in Florida, the sensitivity analysis will examine how the impact will change as the ratio of the labor input requirements increases or decreases.

Another important piece of information is the energy use in deconstruction. According to the review in this dissertation, no previous literature systematically examined the energy use of deconstruction. Thus, this analysis adjusts the energy use coefficients of demolition. The Economic Census 2007 provided on-highway and off-highway diesel fuel use in demolition. The on-highway diesel fuel use in demolition was 1.285 TJ per million dollars of revenue, and the off-highway diesel fuel use was 1.133 TJ per million dollars of revenue in California. The on-highway energy use may decrease when deconstruction diverts the greater amount of CDW and the total travel distance of the CDW decreases. Roughly 10% of total transportation energy use for deconstruction declines compared to that for demolition as it will be demonstrated in next transportation scenario. Thus, the on-highway diesel fuel use of demolition is adjusted by 10%.

Since most building structures are dismantled by workers in deconstruction, deconstruction may considerably save off-highway energy use. However, the extent of energy savings is unknown. This research arbitrarily assumes that 90% of off-highway diesel fuel is saved. Because of great uncertainty in off-highway energy savings, the sensitivity analysis will examine the impact of change in the petroleum-based fuel use coefficient in deconstruction. Through this adjustment, the petroleum-based fuel use coefficient of a deconstruction sector is obtained. In addition, this research assumes that the electricity and natural gas use coefficients are constant in both demolition and deconstruction. The economic and environmental coefficients of demolition and deconstruction are compared in Table 33.

Table 33: Comparison of the Economic and Environmental Coefficients of Demolition and Deconstruction

	Demolition	Deconstruction	Ratio (Deconstruction to Demolition)
Wage and Employment			
Ratio of Labor Cost	-	-	2.09
Average Wage ($)	42,765	34,320	0.80
Income Coefficients	0.257	0.330	1.29
Employment Coefficients	6.010	9.629	1.60
Energy Use Coefficients			
Electricity (Mkwh/$M)	0.047	0.047	1.00
Petroleum-Based Fuel (TJ/$M)	2.637	1.432	0.54
Natural Gas (TJ/$M)	0.128	0.128	1.00
GHG Emission Coefficients			
Petroleum-based Fuel (Ton/$M)	185	100	0.54
Natural Gas (Ton/$M)	6	6	1.00

Transportation Scenario

Deconstruction has an effect on the transportation activities of CDW by altering the required numbers of trips and the transporting paths. In the case of deconstruction, as the amount of CDW transported to mixed CDW recycling facilities and landfills declines, new trips from deconstruction sites to the warehouses of salvaged products are required. The average road distance from a geometric center of a census tract to the nearest warehouse of salvaged products is calculated using the network analyst tool in ArcMap software. Ten warehouses of salvaged products are found in the San Francisco metropolitan area. The average road distance from a geometric center of a census tract to the nearest warehouse is 11.3 miles.

Table 34 shows the weight of CDW shipped during each trip of deconstruction and demolition. When 250 units of residential buildings are deconstructed, roughly 20,000 tons of CDW are generated. Among them, 4,200 tons of salvaged products are shipped to warehouses, which are new trips. The weight of CDW shipped from a census tract to a recycling facility and landfills/transfer stations decreases by 1,200 tons and 3,000 tons, respectively, compared to a demolition scenario Then, the weight of the CDW shipped from the transfer stations/recycling facility to landfills also declines. Consequently, the total weight of the shipped CDW and the total travel distance decrease in the deconstruction scenario by approximately 10%. The total transportation energy use is 0.592 TJ in the case of deconstruction and 0.664 TJ in the case of demolition. As a result, it shows that deconstruction has a positive effect on energy savings in transportation by reducing travel distance and the total CDW shipping weight in the case of the San Francisco metropolitan area.

Table 34: Comparison of Transportation Scenarios of Deconstruction and Demolition

	CDW Shipped (Tons)	Average Distance (Miles)	Total Travel Distance (Miles)	Total Energy Use (TJ)	Total GHG Emissions (Tons)
Deconstruction					
Census Tract- Salvage Store	4,200	11.3	5,927	0.127	9
Census Tract - Recycling facility	13,800	5.8	9,899	0.213	15
Census Tract - Landfills/Transfer Stations	2,000	4.8	1,204	0.026	2
Transfer Stations – Landfills	1,855	17.4	4,024	0.087	6
Recycling facility – Landfills	3450	15.0	6,472	0.139	10
Total	25,305		27,527	0.592	42
Demolition					
Census Tract - Recycling facility	15,000	5.8	10,760	0.231	16
Census Tract - Landfills/Transfer Station	5,000	4.8	3,010	0.065	5
Transfer Station – Landfills	4,637	17.4	10,061	0.216	15
Recycling facility – Landfills	3750	15.0	7,035	0.151	11
Total	28,387		30,865	0.664	47

Simulation Results

The simulation compares the economic and environmental impact of the deconstruction and demolition of 250 units of residential buildings. The results of simulation are summarized in Table 35. The results show that deconstruction has the net positive economic impact in terms of employment and income. If 250 units of residential building are dismantled through the deconstruction technique, the total employment impact is 78 and the total income impact is $3.181 million. By contrast, when the traditional demolition technique applies, the total employment impact is 53 and the total income impact is $2.587 million. Thus, the net job creation is 25 and employees earn an additional $0.594 in wages. The net employment reduction in landfills (-1), service (-2), and mixed CDW recycling (-1) is small.

The results also indicate that deconstruction has a sizeable positive environmental impact on energy savings and GHG emissions reduction. The total energy use of deconstruction is 27.6 TJ whereas the total energy use of demolition is 35.0 TJ. The total net energy savings is 7.5 TJ, consisting of 0.2 TJ in waste collection, 1.1 TJ in landfills, 0.5 TJ in mixed CDW recycling, and 3.9 TJ in deconstruction. The total net GHG reduction of deconstruction is 503 tons. From a spatial perspective, most of the net economic and environmental impact occurs in the San Francisco metropolitan area, and the impact on the rest of the nation is small.

Table 35: Economic and Environmental Impact of Deconstruction in the San Francisco Metropolitan Area

	San Francisco Metropolitan Area					Rest of the Nation					Total				
	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)
Deconstruction															
Primary	0.008	0	0.001	0	0	0.090	0	0.014	1	47	0.097	0	0.015	0.7	48
Utility	0.024	0	0.004	0	17	0.018	0	0.003	1	83	0.042	0	0.006	1.4	100
Construction	0.432	4	0.257	1	78	0.042	1	0.022	0	9	0.474	5	0.279	1.3	87
Manufacturing	0.423	0	0.037	2	121	0.822	2	0.133	3	217	1.245	3	0.170	5.2	338
Waste collection	0.192	1	0.038	1	76	0.031	0	0.009	0	16	0.223	1	0.047	1.3	92
Landfills	0.724	1	0.083	2	132	0.048	0	0.014	0	26	0.772	1	0.097	2.3	158
Service	0.952	6	0.312	2	116	0.678	5	0.213	1	73	1.630	11	0.525	3.4	188
Mixed CDW Recycling	1.401	6	0.268	5	262	0.000	0	0.000	0	0	1.401	6	0.268	4.7	262
Demolition	0.000	0	0.000	0	0	0.000	0	0.000	0	0	0.000	0	0.000	0.0	0
Deconstruction	4.386	52	1.775	7	469	0.000	0	0.000	0	0	4.386	52	1.775	7.3	469
Total	8.542	70	2.775	21	1,271	1.729	9	0.406	6.969	471	10.270	78	3.181	27.6	1,742
Demolition															
Primary	0.011	0	0.001	0	1	0.103	0	0.016	1	52	0.114	0	0.017	0.8	53
Utility	0.030	0	0.005	0	20	0.019	0	0.003	1	90	0.049	0	0.008	1.6	110
Construction	0.423	4	0.252	1	77	0.042	1	0.021	0	9	0.465	5	0.273	1.2	86
Manufacturing	0.551	0	0.041	3	169	0.873	2	0.138	4	230	1.424	3	0.179	6.2	400
Waste collection	0.220	1	0.043	1	87	0.035	0	0.010	0	18	0.256	1	0.054	1.5	105
Landfills	1.104	2	0.127	3	202	0.060	0	0.017	0	32	1.164	2	0.144	3.3	234
Service	1.133	7	0.365	3	134	0.781	6	0.242	1	82	1.914	12	0.608	3.9	216
Mixed CDW Recycling	1.563	6	0.299	5	292	0.000	0	0.000	0	0	1.563	6	0.299	5.3	292
Demolition	3.917	24	1.007	11	750	0.000	0	0.000	0	0	3.917	24	1.007	11.2	750
Deconstruction	0.000	0	0.000	0	0	0.000	0	0.000	0	0	0.000	0	0.000	0.0	0
Total	8.952	44	2.140	27	1,731	1.913	9	0.447	7.633	514	10.865	53	2.587	35.0	2,245

Table 35 continued

	San Francisco Metropolitan Area					Rest of the Nation					Total				
	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)
Net Effect															
Primary	-0.003	0	0.000	0	0	-0.013	0	-0.002	0	-5	-0.016	0	-0.002	-0.1	-5
Utility	-0.006	0	-0.001	0	-3	-0.002	0	0.000	0	-7	-0.007	0	-0.001	-0.2	-10
Construction	0.009	0	0.006	0	2	0.000	0	0.000	0	0	0.009	0	0.006	0.0	2
Manufacturing	-0.127	0	-0.004	-1	-48	-0.051	0	-0.005	0	-14	-0.178	0	-0.009	-1.0	-62
Waste collection	-0.028	0	-0.006	0	-11	-0.004	0	-0.001	0	-2	-0.033	0	-0.007	-0.2	-13
Landfills	-0.380	-1	-0.044	-1	-69	-0.012	0	-0.003	0	-6	-0.392	-1	-0.047	-1.1	-76
Service	-0.181	-1	-0.053	0	-18	-0.103	-1	-0.029	0	-9	-0.284	-2	-0.083	-0.5	-27
Mixed CDW Recycling	-0.163	-1	-0.031	-1	-30	0.000	0	0.000	0	0	-0.163	-1	-0.031	-0.5	-30
Demolition	-3.917	-24	-1.007	-11	-750	0.000	0	0.000	0	0	-3.917	-24	-1.007	-11.2	-750
Deconstruction	4.386	52	1.775	7	469	0.000	0	0.000	0	0	4.386	52	1.775	7.3	469
Total	-0.410	26	0.635	-7	-460	-0.185	-1	-0.041	-0.664	-43	-0.595	25	0.594	-7.5	-503

Sensitivity Analysis

The sensitivity analysis involves changes in labor and energy input ratios and in the number of dismantled housing units. In the initial setting of the simulation, the labor input of deconstruction is 2.09 times as much as that of demolition. The sensitivity analysis examines the change in the economic and environmental impact when the ratio of labor input decreases to 1.80, 1.60, and 1.40. The extent to which employment decreases is associated with the use of mechanical equipment. If deconstruction employs fewer employees, it means that it is likely to utilize more mechanical equipment. Hence, the sensitivity analysis also changes the ratio of the energy use coefficients (deconstruction to demolition) from the initial 0.54 to 0.65, 0.75, and 0.85.

The results of the sensitivity analysis indicate that the magnitude of the positive economic impact decreases considerably, but the magnitude of the positive environmental impact decreases only slightly. The net employment effect declines by 57% from 25 (initial) to 11 (ALT3), and the net income effect also decreases by 81% from \$0.594 million (initial) and \$0.110 million (ALT3). The net energy savings effect decreases by 32% from -7.5 TJ (initial) to -5.1 TJ (ALT3), and the net GHG emissions decrease by 34% from -503 tons (initial) to -332 tons (ALT3).

The sensitivity analysis also investigates how the economic and environmental impact will change when the number of housing units dismantled through the deconstruction technique decreases or increases: 100 units in ALT 4, 400 units in ALT 5, and 550 units in ALT 6. When only 100 units are deconstructed, the net employment effect declines to 10 and the net income effect to \$0.238 million. When deconstructed housing units increase to 400 and 550, the net employment effect rises to 40 and 55. The net income effect increases to $0.951 and $1.307 million. The net positive environmental impact of energy savings and GHG emission reduction also increase proportionally.

Table 36: Results of the Sensitivity Analysis of Deconstruction

		Labor/Energy Input Ratio			Units Demolished		
	Initial	**ALT1**	**ALT2**	**ALT3**	**ALT4**	**ALT5**	**ALT6**
Change of Input Ratio (Deconstruction to Demolition)							
Labor Input Ratio	2.09	1.80	1.60	1.40	-	-	-
Energy Use Coefficients Ratio	0.54	0.65	0.75	0.85	-	-	-
Units Demolished	250	-	-	-	100	400	550
Deconstruction							
Output ($ Millions)	10.270	10.069	9.930	9.791	4.108	16.433	22.595
Employment (Persons)	78	73	68	64	31	126	173
Income ($ Millions)	3.181	2.978	2.837	2.697	1.272	5.090	6.999
Energy (TJ)	27.6	28.4	29.2	30.0	11.0	44.1	60.7
GHG (Tons)	1,742	1,803	1,860	1,913	697	2,787	3,832
Demolition							
Output ($ Millions)	10.865	10.865	10.865	10.865	4.346	17.385	23.904
Employment (Persons)	53	53	53	53	21	85	117
Income ($ Millions)	2.587	2.587	2.587	2.587	1.035	4.139	5.691
Energy (TJ)	35.0	35.0	35.0	35.0	14.0	56.0	77.1
GHG (Tons)	2,245	2,245	2,245	2,245	898	3,592	4,939
Net Effect							
Output ($ Millions)	-0.595	-0.797	-0.935	-1.074	-0.238	-0.952	-1.309
Employment (Persons)	25	19	15	11	10	40	55
Income ($ Millions)	0.594	0.391	0.250	0.110	0.238	0.951	1.307
Energy (TJ)	-7.5	-6.6	-5.8	-5.1	-3.0	-11.9	-16.4
GHG (Tons)	-503	-442	-384	-332	-201	-805	-1,107

7.2. Waste Carpet Recycling

This section analyzes the economic and environmental impact of two cases of waste carpet recycling. The first simulation examines the production of recycled nylon 6. As illustrated in Chapter 3, nylon 6 is the most valuable material in waste carpet, and the production of recycled nylon 6 is more likely to be integrated into the vertical integrated system. Hence, the research posits that a recycling facility producing recycled nylon 6 from waste carpet is located in the Atlanta metropolitan area near to cluster of carpet manufacturing and that the southeastern regional collection systems supply the waste carpet to this recycling facility. This simulation estimates the economic and environmental impact of recycled nylon 6 production and associated transportation activities.

The other case of waste carpet recycling investigated here is the production of recycled carpet padding from waste carpet. A recycling facility for recycled carpet padding production is an example of a small independent recycling business as shown in Chapter 3. Although the Seattle metropolitan area does not have a strong industrial base for carpet manufacturing or recycling, state and local governments have a proactive recycling policy and seek to attract recycling business. Hence, a small independent recycling company may be a suitable business model in the Seattle metropolitan area. The research assumes that a recycling facility equipped with a recycled carpet padding manufacturing process locates in the Seattle metropolitan area, and it serves Washington in terms of waste carpet collection. The second simulation examines the extent of the economic and environmental impact when waste carpet is diverted and collected in Washington and recycled into carpet padding products in a facility located in the Seattle metropolitan area.

7.2.1. Recycled Nylon 6 Production in the Atlanta Metropolitan Area

Scenario of Recycling and Disposal Paths

This simulation compares a recycling path and a disposal path pertaining to the recycling of nylon 6 fiber in waste carpet as illustrated in Figure 20. The basic recycling scenario is that 100 million pounds of waste carpet is diverted and collected in the southeastern states and processed in a recycling facility located in the Atlanta metropolitan area. The recycling facility produces approximately 30 million pounds of recycled nylon 6 fiber. Two major industrial activities take place in this recycling path. One is regional-scale transportation activity, and the other is recycled nylon 6 production activity. Thus, the recycling path evaluates the economic and environmental impact of transporting and recycling activities pertaining to waste nylon 6.

The simulation also accounts for industrial activities that will be displaced by recycling industrial activities. The disposal path includes three displaced industrial activities: the transportation and disposal of waste carpet to landfills and the production of virgin nylon 6. It assumes that recycling substitutes for the import of the virgin nylon 6 produced in the rest of the nation. The economic and environmental effects of the industrial activities in the disposal path are separately computed and then subtracted from those of recycling activities to obtain the net economic and environmental impact.

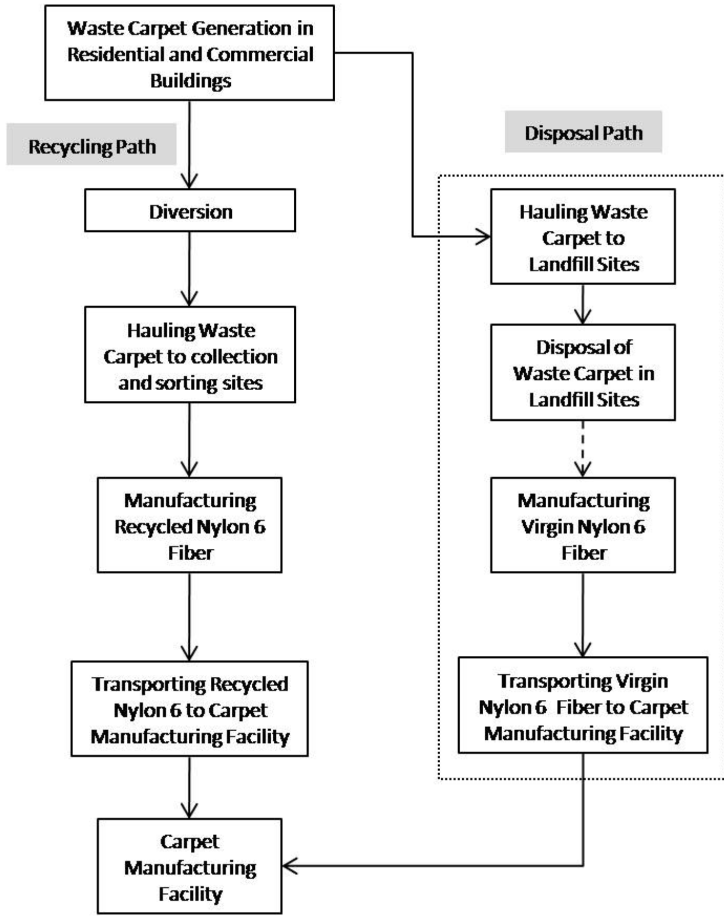

Figure 20: Recycling and Disposal Paths of Nylon 6 Materials in Waste Carpet

Recycled Nylon 6 Production

Since no industry sector in the IO model represents a recycled nylon 6 production process, the research constructed the economic and environmental profile of this recycling facility based on several data sources. The input structure of the recycling process is obtained from the previous research on engineering modeling (Subbiah, 2008), the energy use of the recycling process from an expert (Realff, Personal communication, 2011), employment, wages, and the price of recycled materials from the carpet recycling survey.

The recycling facility has a recycling capacity to process approximately 100 million pounds of waste carpet and manufacture 30 million pounds of recycled nylon 6 fiber per year. The electricity and natural gas is primary energy source. The nylon 6 recycling process consumes 0.120Mkwh of electricity per million dollars of revenue and 13.958 TJ of natural gas per million dollars of revenue. It also emits 735 tons of GHGs per $million revenues from direct combustion of natural gas. It hires roughly 2.00 employees per million dollars of revenue; the positions include equipment operators, manual workers, drivers, administrators, sales personnel, researchers, and managers. Their average incomes are $51,000 per year.

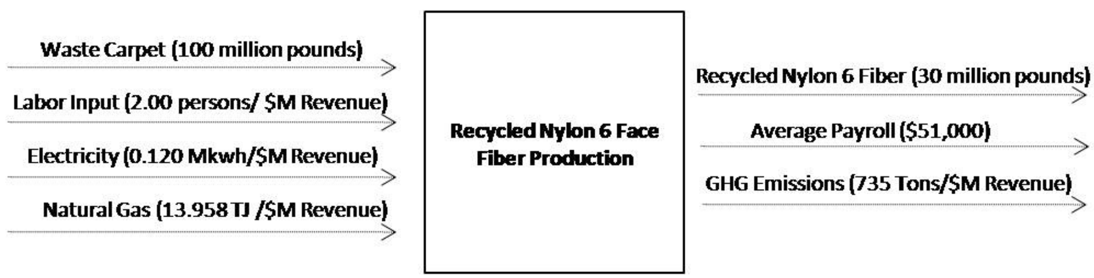

Figure 21: Input and Output Flow of Recycled Nylon 6 Face Fiber Production

The economic and environmental profile of the production of recycled nylon 6 needs to be compared with the production of virgin nylon 6 in terms of their economic and environmental coefficients. With respect to environmental coefficients, previous studies have indicated that the production of virgin caprolactam and nylon 6 fiber is a more energy intensive process than that of recycled caprolactam or recycled nylon 6 (Binder et al, 2010; U.S. EPA, 2012). Energy consumption normalized by revenue from the production of virgin nylon 6 fiber is approximately 1.7 times as much as that of recycled nylon 6 fiber.

The economic input coefficients such as intermediate input, employment, and wage for 509 industry sectors are available in IMPLAN. IMPLAN 155 non-cellulosic organic fiber manufacturing is an industry sector in which the major products are synthetic fibers (i.e., nylon, polyolefin, and polyester).[36] Since this sector manufactures multiple types of synthetic fibers, they do not exclusively represent the economic input structure of the production of virgin nylon 6 fiber. Thus, in this simulation, the three economic input coefficients of the IMPLAN 155 sector are modified for the purpose of comparing the production of virgin nylon 6 fiber with the production of recycled nylon 6 face fiber.

The basic assumption is that the production of virgin nylon 6 fiber operates on a larger scale and is a more capital intensive process compared to the production of recycled nylon 6 fiber, suggesting that the number of employees per output is likely to be lower, but the average wages are likely to be higher in virgin nylon 6 production. The average production cost per output of virgin nylon 6 face fiber is likely to be lower than that of recycled nylon 6 face fiber. Hence, this research initially set the ratio (of virgin to recycled fiber) of the wage rate at 1.1, the ratio of employment coefficients at 0.9, and the ratio of the primary input at 0.8.[37] Because some economic parameters are adjusted based on these arbitrary values, the sensitivity analysis will examine these parameters.

---

[36] NAICS 325222,Noncellulosic organic fiber manufacturing is defined as "this U.S. industry consists of establishments primarily engaged in (1) manufacturing noncellulosic (i.e., nylon, polyolefin, and polyester) fibers and filaments in the form of monofilament, filament yarn, staple, or tow, or (2) manufacturing and texturizing noncellulosic fibers and filaments."Accessed at a website of the U.S. Census Bureau, http://www.census.gov/econ/industry/def/d325222.htm

[37] IMPLAN 155 noncellulosic organic fiber manufacturing purchases the feedstock of nylon from the IMPLAN 151 Other basic organic chemical manufacturing and IMPLAN 152 Plastics material and resin manufacturing.

Regional Collection System

The simulation assumes that a recycling facility can process roughly 100 million pounds of waste carpet, which suggests the need for a regional-scale collection system in a current low diversion rate. According to the study that estimated the waste carpet disposal rate (Ai, Personal communication, 2010), the range of waste carpet disposal rates of the Atlanta metropolitan area is 18 to 27 pounds/person/year. If the mean value, 22.5 pounds/person/year and the diversion rate of 10% apply, a population of 44 million would be required to supply waste carpet to a recycling facility. Since the population of the southeastern states in 2010 was 56 million,[38] a southeastern-scale collection system could meet the demand of a recycling facility.

For simplicity of the collection scenario, two assumptions are made. One is that waste carpet disposal and diversion rates are constant across all counties in the southeastern states, so the diverted amount of waste carpet is proportional to the population of the county. The second assumption is that all diverted waste carpet is hauled by end-users to the drop-off site located in a geometric center of each county. All industrial transportation activities begin at each drop-off site. The total amount of diverted waste carpet in each county is calculated by multiplying the mean value of diverted waste carpet per person by the population of the county.

The transportation of the disposal path in the collection scenario is simple. Waste carpet is transported from a drop-off site to the nearest landfills. Alternatively, the recycling path consists of two transport steps. First, the waste carpet at the drop-off sites is hauled to regional warehouses, and then the collected waste carpet is hauled to a

[38] The southeastern states include Alabama, Florida, Georgia, Mississippi, North Carolina, South Carolina, and Tennessee.

recycling facility in the Atlanta metropolitan area. The location of 15 regional warehouses in Shaw's reverse collection system is used in this transportation scenario. The location of landfills and regional warehouses in the southeastern states is displayed in Figure 22.

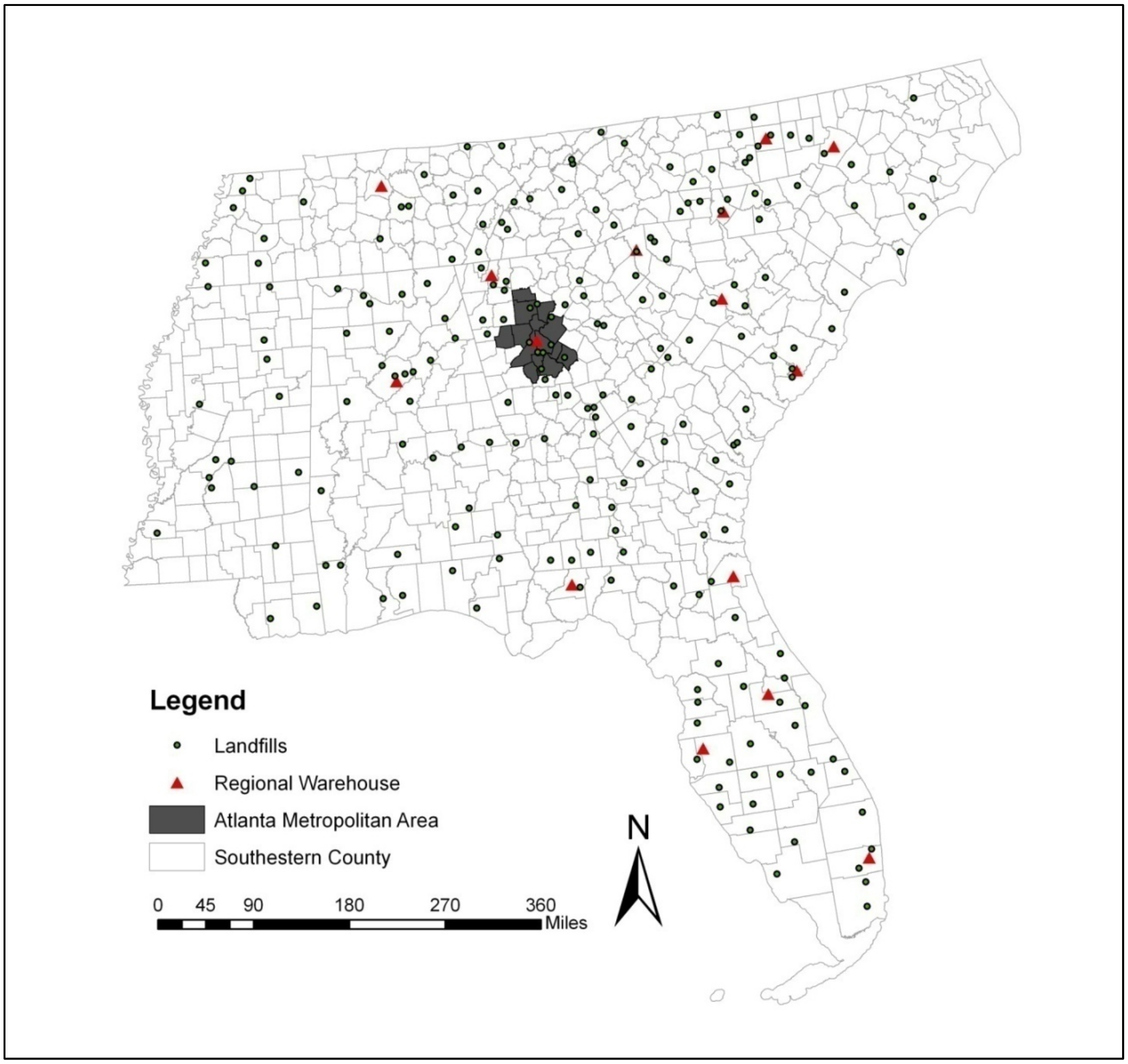

Figure 22: Location of Landfills and Warehouses in the Southeastern States

Based on the location information and a highway planning network map, the three types of road distances are calculated using the network analyst tool in ArcMap software:

1) the distance from county drop-off sites to the nearest landfills, 2) the distance from county drop-off sites to the nearest regional warehouses, and 3) the distance from regional warehouses to a recycling facility. For estimating the total travel distance and the associated energy use, the research assumes that a six-ton roll-off bin and a 6.8 mpg heavy duty diesel vehicle (HDDV) 8A are utilized. According to the carpet recycling survey, the average collection fee is roughly $0.05 per pound. The same fee for collection applies to both recycling and disposal paths in the calculation of the total revenue. Finally, energy use per million dollars of revenue and GHG emissions per million dollars of revenue are calculated, shown in Table 37. The total travel distance of the recycling path is significantly longer than that of the disposal path. More than 15 times as much energy and GHGs are consumed and emitted in the recycling path as in the disposal path.

Table 37: Comparison of the Collection System for the Recycling and Disposal Paths

	Waste Carpet Collected (Tons)	Total Travel Distance (Miles)	Total Revenue ($M)	Total Energy Use (TJ)	Total GHG Emissions (Tons)	Energy Use per Revenue (TJ/$M)	GHG Emissions per Revenue (Tons/$M)
Recycling Path							
Atlanta Metropolitan Area	5,840	52,880	0.644	1.14	80	1.77	124
Rest of Southeastern States	40,045	3,329,468	4.414	71.59	5,023	16.22	1,138
Sum	45,885	3,382,348	5.058	72.72	5,102	14.38	1,009
Disposal Path							
Atlanta Metropolitan area	3,731	11,216	0.411	0.24	17	0.59	41
Rest of Southeastern States	42,154	211,408	4.647	4.55	319	0.98	69
Sum	45,885	200,191	5.058	4.79	336	0.95	66

Simulation Results

The simulation evaluates the economic and environmental impact of industrial activities in the recycling and disposal paths. The final demand term is the output of each industrial activity performed in the recycling and disposal paths. The effect of the recycling path, the effect of a path disposal, and the net effect are displayed in Table 38.

The diversion and recycling of nylon 6 materials create a net positive economic impact in terms of output, employment, and income. They also result in a reduction in energy use and GHG emissions. The magnitude of the net positive effect is relatively small: an output of $5.383 million, the addition of 20 employees, and an income of $0.957 million. However, with regard to the total environmental net effect, the reduction in energy use and GHG emissions are noticeable: a reduction of 369 TJ in energy use and 14,252 tons in GHG emissions. These figures translate into energy savings of 12,398 MJ per 1,000 pounds of recycled nylon 6 products, and GHG emission reduction of 0.479 tons per 1,000 pounds of recycled nylon 6 products. The higher energy intensity of the production of virgin nylon 6 significantly contributes to the net positive energy and environmental impact. However, the negative environmental impact from longer hauling distance of diverted waste carpet is sizable: an additional 83 TJ of net energy use in transportation and an additional 6,096 tons of net GHG emissions; nevertheless, these effects are offset by the positive effect of nylon 6 recycling.

From a spatial perspective, the results shows that the net positive economic and negative environmental effects concentrate in the Atlanta metropolitan area mainly due to an assumption of replacing the imported virgin nylon 6 produced in the rest of the nation. A nylon 6 recycling facility could create the direct and indirect 274 new jobs that pay

$14,224 million in wages, but it would entail the consumption of 1,325 TJ in energy and emissions of 78,256 tons of GHGs in the Atlanta metropolitan area. Conversely, in the rest of the nation, 253 jobs would disappear, 1,694 TJ of energy would be saved, and 92,509 tons of GHG emissions would be reduced as a result of displaced industrial activities.

Table 38: Economic and Environmental Impact of the Production of Recycled Nylon 6 Face Fiber

	Atlanta Metropolitan Area					Rest of the Nation					Total				
	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)
Recycling Path															
Primary	0.100	0	0.001	0	5	2	4	0	3	147	1.878	4	0.202	3	152
Utility	7.172	8	0.756	244	20,131	1	1	0	30	2,264	8.184	9	0.887	274	22,395
Construction	0.303	3	0.096	1	54	0	2	0	0	32	0.482	4	0.156	1	86
Manufacturing	1.198	3	0.149	4	219	8	12	1	38	2,388	9.397	15	0.989	41	2,607
Virgin Nylon 6 Face Fiber Manufacturing	0.000	0	0.000	0	0	0	0	0	2	85	0.055	0	0.006	2	85
Waste Carpet Collection	6.409	49	2.083	11	794	5	39	1	82	5,725	11.441	89	3.543	93	6,519
Landfills	0.875	5	0.278	8	549	0	2	0	3	227	1.297	7	0.398	11	775
Service	9.637	65	3.592	15	820	10	65	3	18	1,080	19.337	130	6.784	33	1,901
Recycled Nylon 6 Face Fiber Manufacturing	74.405	148	7.614	1,047	56,026	0	0	0	0	0	74.405	148	7.614	1,047	56,026
Total	100.099	281	14.568	1,330	78,599	26	126	6	175	11,948	126.475	406	20.578	1,505	90,547
Disposal Path															
Primary	0.001	0	0.000	0	0	4	11	0	5	267	3.806	11	0.410	5	267
Utility	0.050	0	0.005	2	186	9	11	1	250	18,641	8.879	11	1.134	252	18,827
Construction	0.003	0	0.001	0	0	0	4	0	1	77	0.433	4	0.150	1	77
Manufacturing	0.080	0	0.010	0	22	32	77	4	115	6,517	31.731	77	3.847	115	6,539
Virgin Nylon 6 Face Fiber Manufacturing	0.001	0	0.000	0	1	47	84	5	1,419	73,701	47.166	84	4.778	1,419	73,702
Waste Carpet Collection	0.461	4	0.150	0	19	6	46	2	10	404	6.353	50	1.859	10	423
Landfills	0.125	1	0.040	1	78	2	13	1	18	1,271	2.489	14	0.714	19	1,349
Service	0.399	3	0.137	1	36	20	132	7	52	3,577	20.227	135	6.728	52	3,613
Recycled Nylon 6 Face Fiber Manufacturing	0.007	0	0.001	0	0	0	0	0	0	0	0.007	0	0.001	0	0
Total	1.127	7	0.344	5	342	120	379	19	1,870	104,457	121.092	386	19.620	1,874	104,799

Table 38 Continued

	Atlanta Metropolitan Area					Rest of the Nation					Total				
	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)
Net Effect															
Primary	0.099	0	0.000	0	5	-2.026	-7	-0.208	-2	-120	-1.927	-7	-0.207	-2	-115
Utility	7.122	8	0.750	242	19,946	-7.817	-10	-0.997	-219	-16,377	-0.696	-2	-0.247	23	3,568
Construction	0.300	3	0.095	1	54	-0.251	-2	-0.089	-1	-45	0.049	0	0.006	0	8
Manufacturing	1.119	2	0.139	4	197	-23.452	-64	-2.997	-78	-4,129	-22.334	-62	-2.858	-74	-3,932
Virgin Nylon 6 Face Fiber Manufacturing	-0.001	0	0.000	0	-1	-47.111	-84	-4.773	-1,417	-73,616	-47.111	-84	-4.773	-1,418	-73,617
Waste Carpet Collection	5.948	46	1.934	11	775	-0.861	-7	-0.250	72	5,321	5.087	39	1.684	83	6,096
Landfills	0.750	4	0.238	7	470	-1.943	-11	-0.554	-15	-1,044	-1.193	-7	-0.316	-8	-574
Service	9.237	63	3.455	14	785	-10.127	-67	-3.399	-34	-2,497	-0.890	-5	0.056	-20	-1,712
Recycled Nylon 6 Face Fiber Manufacturing	74.398	148	7.613	1,047	56,026	0.000	0	0.000	0	0	74.398	148	7.613	1,047	56,026
Total	98.972	274	14.224	1,325	78,256	-93.589	-253	-13.267	-1,694	-92,509	5.383	20	0.957	-369	-14,252

Sensitive Analysis

The sensitivity analysis examines changes in the economic input coefficients and the substitution rate of recycled nylon 6. First, as discussed above, the economic input coefficients in the comparison of virgin and recycled nylon 6 fiber production processes involve some uncertainty. Three parameters are under examination in the sensitivity analysis: employment coefficients, average wages, and primary input coefficients. The ratios of those coefficients are adjusted as displayed in Table 39.

In ALT1, assuming that the wage rates and the employment per output of virgin and recycled nylon 6 fiber are the same, the impact of the positive net employment and income decrease to 10 and $0.836 million, respectively. By contrast, in ALT2 and ALT3, assuming that the recycled nylon 6 production process employs greater numbers of workers at lower wages, the impact of the positive net employment and income increase to 30 and $1.102 million and 40 and $1.416 million, respectively. Approximately every 10% change in the ratios of the wage and employment coefficients leads to an increase or a decrease of 10 employees.

The sensitivity analysis also investigates changes in economic and environmental impact when the substitution rate (i.e., the recycled nylon 6 displaces the virgin nylon 6) decreases, suggesting that the supply of recycled nylon 6 creates a new market outlet. The sensitivity analysis investigated 80%, 60%, and 40% substitution rates.

When the substitution rate is 80%, meaning 20% of recycled nylon 6 fiber is sold to novel outlets, 44 new jobs are created and $2.223 million in income are added to the economy. When the substitution rate is 40%, they increase to 132 and$7.069 million; that is, roughly, a 10% decrease in the substitution rate leads to an increase of 20 employees.

However, the positive environmental impact of reduced energy use and GHG emissions disappears. Approximately, such effect of reductions in energy use and GHG emissions will disappear at about a 69% and 76% substitution rate, respectively.

Table 39: Results of the Sensitivity Analysis of Recycled Nylon 6 Face Fiber Production

		Economic Input Coefficients			Substitution Rate		
	Initial	**ALT1**	**ALT2**	**ALT3**	**ALT4**	**ALT5**	**ALT6**
Ratio (Virgin to Recycled)							
Employment Coefficients	0.9	1.0	0.8	0.7	-	-	-
Average Wage	1.1	1.0	1.2	1.3	-	-	-
Primary Input	0.8	1.0	0.8	0.7	-	-	-
Substitution	100%	-	-	-	80%	60%	40%
Price Ratio	0.8	0.8	0.8	0.8	0.9	0.95	1.0
Recycling Path							
Output ($ Millions)	126.475	126.476	126.475	126.475	113.495	108.030	103.111
Employment (Persons)	406	406	406	406	368	352	338
Income ($ Millions)	20.578	20.578	20.577	20.577	18.585	17.746	16.991
Energy (TJ)	1,505	1,505	1,505	1,505	1,349	1,282	1,223
GHG (Tons)	90,547	90,547	90,547	90,547	81,208	77,275	73,736
Disposal Path							
Output ($ Millions)	121.092	121.754	121.092	120.430	99.351	78.006	56.397
Employment (Persons)	386	397	377	366	324	266	206
Income ($ Millions)	19.620	19.742	19.475	19.161	16.361	13.168	9.922
Energy (TJ)	1,874	1,878	1,874	1,870	1,507	1,143	776
GHG (Tons)	104,799	105,033	104,799	104,565	84,336	64,014	43,598
Net Effect							
Output ($ Millions)	5.383	4.722	5.383	6.044	14.144	30.023	46.714
Employment (Persons)	20	10	30	40	44	86	132
Income ($ Millions)	0.957	0.836	1.102	1.416	2.223	4.578	7.069
Energy (TJ)	-369	-373	-369	-365	-159	140	447
GHG (Tons)	-14,252	-14,486	-14,252	-14,019	-3,129	13,261	30,138

7.2.2. Recycled Carpet Padding Production in the Seattle Metropolitan Area

Scenario of Recycling and Disposal Paths

Waste carpet is currently recycled as a carpet padding product, illustrated in Chapter 3. This simulation presents a hypothetical recycling facility that produces recycled carpet padding products located in the Seattle metropolitan area. The hypothetical facility has a capacity to process roughly 25 million pounds of waste carpet per year and to manufacture approximately 10 million pounds of recycled carpet padding products. The recycling and disposal paths are compared in Figure 23. The recycling path includes two primary industrial activities: transporting diverted waste carpet to a recycling facility and manufacturing recycled carpet padding products. The simulation takes into account industrial activities that may be displaced when waste carpet is recycled. The disposal path includes transporting waste carpet to landfills, disposing waste carpet in landfills, and manufacturing other types of carpet padding products. This simulation assumes that recycled carpet padding products substitute for the polyurethane foam carpet padding, the dominant type of carpet padding in the market.[39]

[39] The Carpet Cushion Council estimated that 89% of domestically produced carpet padding consists of bonded foam. Accessed from the website of the Carpet Cushion Council on April 8, 2012, http://www.carpetcushion.org/recycling.cfm

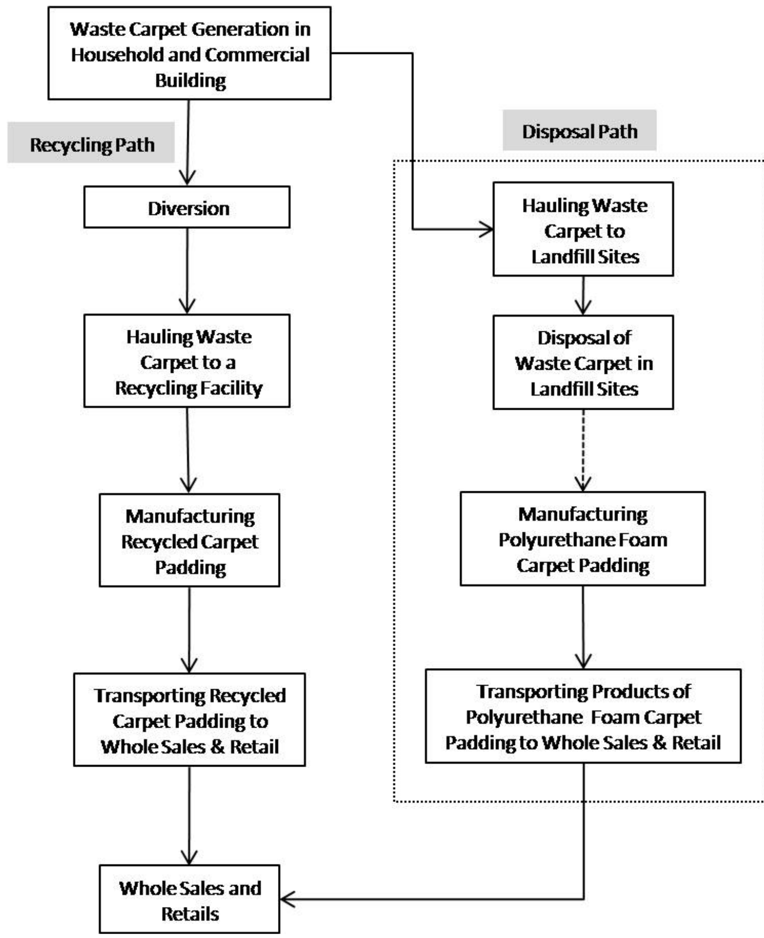

Figure 23: Recycling and Disposal Paths of Carpet Padding Production

Production of Recycled Carpet Padding

For creating a new sector for the production of recycled carpet padding in the regional environmental IO model, this research builds an economic and environmental profile of a recycling facility based on several data sources: the economic input structure and processing capacity from a previous study of an engineering model (Subbiah, 2008), the employment and wage rates from a carpet recycling survey, and energy consumption information from an expert (Realff, Personal communication, 2011). As shown in Figure 24, the recycling facility can process roughly 25 million pounds of waste carpet per year and produce 10 million pounds of recycled carpet padding products. Operation of this facility requires 2.14 employees per million dollars of revenue. The average wage per employee is about $40,000. Compared to the nylon 6 recycling process, the recycled carpet padding process operates on a small scale, involves slightly higher labor intensity, but pays lower wages. This process consumes 0.134 Mkwh of electricity per million dollars of revenue and 0.169 TJ of natural gas per million dollars of revenue. This process requires considerably lower energy intensity than the nylon 6 recycling process. It also emits only 8 tons of GHGs per million dollars of revenue from the direct combustion of natural gas.

Figure 24: Input and Output Flow of Recycled Carpet Padding Production

The economic and environmental profile of the manufacturing process of recycled carpet padding is compared with that of polyurethane foam carpet padding products. Although no sector of the IO model specifically represents the production of polyurethane foam carpet padding because of the aggregated industry classification in IMPLAN, in this analysis, IMPLAN 178, the foam product manufacturing sector, is assumed to approximate the production of polyurethane foam carpet padding. Data pertaining to the average wage rate and input structure are directly retrieved from IMPLAN, and the energy consumption, and GHG emissions per output are obtained from the Seattle-metropolitan environmental accounts established in Chapter 5. To calculate employment per output, the research investigated the members of the Carpet Cushion Council through the business database of Reference U.S.A. In the case of small-size companies that produce polyurethane foam carpet padding, the employment per output ranged from 1.8 to 1.9 per million dollars. As a result, two production processes are briefly compared in Table 40. The table shows that the recycled carpet padding process consumes less energy but hires slightly more employees than the polyurethane foam carpet padding process. The ratios of the wage rate and employment coefficients are scrutinized in the sensitivity analysis because of uncertainty relating to the data sources.

Table 40: Comparison of Two Carpet Padding Production Systems

	Polyurethane Form Carpet Padding	**Recycled Carpet Padding**	**Ratio**
Average Wage Per Employee ($)	46,089	40,078	1.15
Employment Coefficients (employees per $M revenues)	1.924	2.138	0.90
Total Energy Use (TJ/$M)	1.006	0.658	1.53

Collection System

The recycling facility in the Seattle metropolitan area is assumed to accept waste carpet generated in the state of Washington. According to a study that estimated the waste carpet disposal rate (Ai, Personal communication, 2010), the range of the waste carpet disposal rate in the Seattle metropolitan area is from 19 to 29 pounds/person/year. If the mean value, 24.0 pounds/person/year, applies, roughly a 17% diversion rate in Washington is required to supply waste carpet as input material to the recycled carpet padding production facility.

For simplicity of waste carpet collection systems, two assumptions are made. The first is that waste carpet disposal and diversion rates are constant across the counties in Washington, so the amount of collected waste carpet is proportional to the population of each county. The second assumption is that the diverted waste carpet is transported to drop-off sites located in a geometric center of each county by end-users and that all industrial transportation activities start from each drop-off site.

The disposal path consists of two trips. Waste carpet in drop-off sites is first sent to the nearest landfills or transfer stations. If it is hauled to transfer stations, it is then hauled to the nearest landfills. By contrast, the recycling path consists of only a single trip. Waste carpet at drop-off sites is directly shipped to a recycling facility. The locations of landfills and transfer stations in Washington are shown in Figure 25. Three types of road distances are calculated using the network analyst tool in ArcMap software: 1) the distance from county drop-off sites to the nearest transfer station or landfills, 2) the distance from transfer stations to the nearest landfills, and 3) the distance from county drop-off sites to a recycling facility.

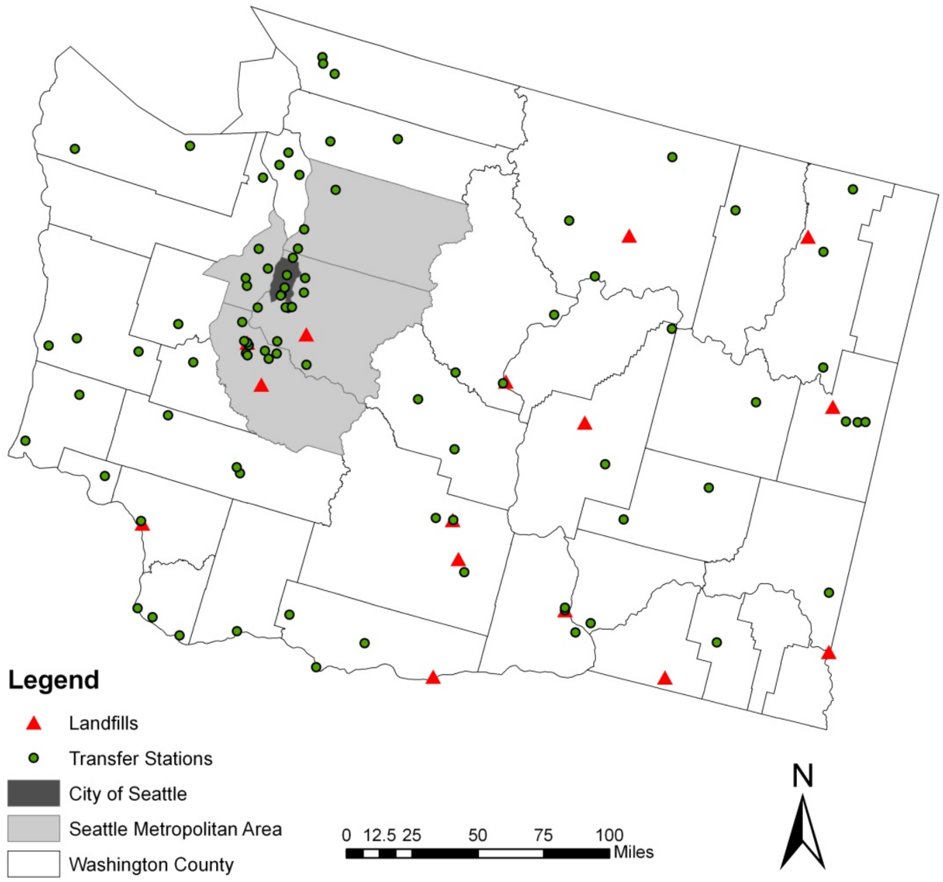

Figure 25: Location of Landfills and Transfer Stations in Washington

To determine the total number of trips and the total travel distance, this study used the same bin size (6 tons) and vehicle type (HDDV8A) as it did in the case of nylon 6 recycling. The total travel distance, energy use, and GHG emissions from the transportation of waste carpet in both the recycling and disposal paths are displayed in Table 41.The total travel distance of the recycling path is nearly twice as long as that of the disposal path. As a result, the energy consumption and GHG emissions of the recycling path are considerably greater rather than those of the disposal path.

Table 41: Comparison of the Recycling and Disposal Paths in Recycled Carpet Padding Production

	Waste Carpet Collected (Tons)	Total Travel Distance (Miles)	Total Revenue ($M)	Total Energy Use (TJ)	Total GHG Emissions (Tons)	Energy Use per Revenue (TJ/$M)	GHG Emissions per Revenue (Tons/$M)
Recycling path							
Seattle metro	6,753	51,979	0.744	1.12	78	1.50	105
Rest of Washington State	5,581	196,349	0.615	4.22	296	6.86	481
Sum	12,333	248,328	1.359	5.34	375	3.93	276
Disposal path							
Seattle metro	6,753	68,275	0.744	1.47	103	1.97	138
Rest of Washington State	5,581	59,470	0.615	1.28	90	2.08	146
Sum	12,333	127,745	1.359	2.75	193	2.02	142

Simulation Results

The economic and environmental impact of the production of recycled carpet padding is analyzed in the two-region environmental IO model, and the effects of the recycling and disposal paths are compared in Table 42. As a whole, the result shows that the extent of the economic impact of recycling and that of the disposal paths are similar. It displays that the net effect on employment is very small but positive (1 employee) and on income, it is small but negative (-$0.299 million). The positive economic effects in waste carpet collection and recycled carpet padding production are mostly offset by the impact of displaced industrial activities such as polyurethane foam carpet padding production, landfills, and other manufacturing and service sectors. With regard to the environmental impact, the recycling path yields relatively small reductions in energy use (42 TJ) and GHG emissions (2,955 tons).

With respect to the geographical perspective, under the assumption that recycled carpet padding displaces imported polyurethane foam carpet padding products, the diversion and recycling of waste carpet into recycled carpet padding provides new economic opportunities for the Seattle metropolitan area. It adds $15.268 million in output, 43 jobs, and 1.798 million dollars of income. However, it also generates additional environmental burden of 13 TJ of energy use and 343 tons of GHG emissions on the Seattle metropolitan area.

Table 42: Economic and Environmental Impact of Recycled Carpet Padding Production

	Seattle Metropolitan Area					Rest of the Nation					Total				
	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)	Output ($M)	Employment (Persons)	Income ($M)	Energy (TJ)	GHG (Tons)
Recycling Path															
Primary	0.007	0	0.000	0	1	0.198	0	0.022	0	16	0.205	1	0.022	0	17
Utility	0.029	0	0.003	0	24	0.050	0	0.007	3	216	0.079	0	0.010	3	240
Construction	0.041	0	0.015	0	7	0.016	0	0.006	0	3	0.057	0	0.021	0	10
Manufacturing	0.261	1	0.030	1	54	1.239	2	0.139	6	356	1.499	3	0.169	6	410
PU Foam Padding	0.006	0	0.001	0	0	0.004	0	0.000	0	0	0.010	0	0.001	0	1
Waste Carpet Collection	1.939	14	0.584	3	204	0.984	8	0.286	7	474	2.923	22	0.870	10	678
Landfills	0.145	1	0.046	0	29	0.134	1	0.038	0	28	0.279	2	0.084	1	57
Service	1.762	11	0.612	4	255	1.722	12	0.565	3	181	3.483	23	1.176	7	436
Recycled Carpet Padding	13.095	28	1.122	9	111	0.000	0	0.000	0	0	13.095	28	1.122	9	111
Total	17.285	55	2.413	18	685	4.347	23	1.063	19	1,273	21.631	79	3.475	36	1,958
Disposal Path															
Primary	0.004	0	0.000	0	0	0.407	1	0.044	1	32	0.411	1	0.044	1	32
Utility	0.006	0	0.001	0	3	0.247	0	0.037	17	1,315	0.253	0	0.037	17	1,318
Construction	0.004	0	0.001	0	1	0.057	0	0.019	0	10	0.061	1	0.020	0	11
Manufacturing	0.092	0	0.005	0	31	4.025	5	0.369	22	1,356	4.117	5	0.374	23	1,387
PU Foam Padding	0.000	0	0.000	0	0	12.774	25	1.133	21	1,009	12.774	25	1.133	21	1,009
Waste Carpet Collection	0.790	6	0.238	2	109	0.761	6	0.221	2	111	1.552	12	0.459	3	220
Landfills	0.775	4	0.243	2	153	0.779	4	0.222	2	163	1.553	9	0.465	5	316
Service	0.346	2	0.126	1	44	3.407	23	1.115	8	576	3.754	26	1.241	9	620
Recycled Carpet Padding	0.000	0	0.000	0	0	0.000	0	0.000	0	0	0.000	0	0.000	0	0
Total	2.016	13	0.615	5	343	22.458	65	3.159	73	4,571	24.475	78	3.774	78	4,913

Table 42 Continued

	Seattle Metropolitan Area					Rest of the Nation					Total				
	Output	Employment	Income	Energy	GHG	Output	Employment	Income	Energy	GHG	Output	Employment	Income	Energy	GHG
	($M)	(Persons)	($M)	(TJ)	(Tons)	($M)	(Persons)	($M)	(TJ)	(Tons)	($M)	(Persons)	($M)	(TJ)	(Tons)
Net Effect															
Primary	0.003	0	0.000	0	0	-0.209	-1	-0.022	0	-16	-0.206	-1	-0.022	0	-15
Utility	0.023	0	0.002	0	21	-0.197	0	-0.030	-14	-1,099	-0.174	0	-0.027	-14	-1,078
Construction	0.037	0	0.014	0	6	-0.042	0	-0.013	0	-7	-0.004	0	0.001	0	-1
Manufacturing	0.169	0	0.025	0	23	-2.786	-3	-0.230	-17	-1,000	-2.618	-3	-0.205	-16	-977
PU Foam Padding	0.006	0	0.001	0	0	-12.770	-25	-1.133	-21	-1,009	-12.764	-25	-1.132	-21	-1,009
Waste Carpet Collection	1.148	8	0.346	1	95	0.223	2	0.065	5	363	1.371	10	0.410	7	458
Landfills	-0.630	-3	-0.198	-2	-125	-0.644	-4	-0.184	-2	-135	-1.274	-7	-0.381	-4	-259
Service	1.415	9	0.485	3	211	-1.685	-11	-0.550	-5	-395	-0.270	-2	-0.065	-2	-184
Recycled Carpet Padding	13.095	28	1.122	9	111	0.000	0	0.000	0	0	13.095	28	1.122	9	111
Total	15.268	43	1.798	13	343	-18.112	-42	-2.097	-54	-3,298	-2.843	1	-0.299	-42	-2,955

Sensitivity Analysis

Sensitivity analysis examines changes in the economic input coefficients and the substitution rates. As the average wage and employment rates involve some uncertainty, the ratios of the wage rate and employment coefficients of polyurethane form carpet padding production to recycled carpet padding production are adjusted. In the ALT1, ALT2, and ALT3, the research assumed that the production of recycling carpet padding was more labor intensive and lower paid compared to that of polyurethane foam carpet padding. This assumption suggests that the employment ratio (polyurethane foam carpet padding to recycled carpet padding) becomes smaller, and the wage ratio becomes larger. The results of the analysis show that the net impact on employment is 4 workers and the net impact on income is -$0.232 million in the ALT3. If roughly 10% of the ratio of employment coefficients change, the impact on employment will increase or decrease by 3. Changes in the ratios of the employment coefficients and average wage result in relatively small changes in the economic indicators.

The sensitivity analysis also investigates the impact of changes in the substitution rates. At a 40% substitution rate, the net employment effect increases to 28, but the energy savings effect decreases to 6TJ. A 10% reduction in the substitution rate leads to an increase of approximately 5 employees.

Table 43: Results of the Sensitivity Analysis of Recycled Carpet Padding Production

	Initial Model	Economic Input Coefficients			Substitution Rate		
		ALT1	ALT2	ALT3	ALT4	ALT5	ALT6
Ratio*							
Employment Coefficient	0.9	0.95	0.85	0.8	-	-	-
Average wage	1.15	1.05	1.2	1.25	-	-	
Substitution	100%	-	-	-	80%	60%	40%
Recycling Path							
Output ($ Millions)	21.631	21.631	21.631	21.631	21.631	21.631	21.631
Employment (Persons)	79	79	79	79	79	79	79
Income ($ Millions)	3.475	3.475	3.475	3.475	3.475	3.475	3.475
Energy (TJ)	36	36	36	36	36	36	36
GHG (Tons)	1,958	1,958	1,958	1,958	1,958	1,958	1,958
Disposal Path							
Output ($ Millions)	24.475	24.475	24.336	24.197	21.281	17.689	13.697
Employment (Persons)	78	79	76	75	70	60	50
Income ($ Millions)	3.774	3.733	3.743	3.707	3.368	2.911	2.403
Energy (TJ)	78	78	77	77	68	56	43
GHG (Tons)	4,913	4,913	4,868	4,822	4,265	3,537	2,727
Net Effect							
Output ($ Millions)	-2.843	-2.843	-2.705	-2.566	0.350	3.943	7.934
Employment (Persons)	1	-1	2	4	9	18	28
Income ($ Millions)	-0.299	-0.258	-0.268	-0.232	0.107	0.564	1.072
Energy (TJ)	-42	-42	-41	-40	-31	-19	-6
GHG (Tons)	-2,955	-2,955	-2,909	-2,864	-2,307	-1,578	-768

*Ratio: Polyurethane foam carpet padding production to recycled carpet padding production

7.3. Conclusion

The simulation demonstrated that the regional environmental IO model is a useful tool for analyzing the economic and environmental impact in regional contexts. The four cases of simulation showed that the product-mix issue associated with the aggregated industry classification of the conventional IO model is critical in environmental IO modeling. It is common that a product or a production system investigated in environmental modeling is more specific than a classified industry sector in the IO model. The disaggregation or the addition of a new sector is necessary to improving the accuracy of estimation. In this simulation, new sectors for mixed CDW recycling, deconstruction, nylon 6 recycling, and recycled carpet padding production were added to existing regional environmental IO models.

The creation of a new sector in environmental IO modeling required an understanding of the physical and economic sides of the recycling facility operation and a compilation of extensive sets of data, including the quantity of input material and energy, products, wages, number of employees, the prices of products, and the structure of economic input. Since no single source typically provides such comprehensive data, this study compiled relevant data from diverse sources such as business surveys, personal communications with businesses and experts in the field, and secondary documents.

The case of mixed CDW recycling in the San Francisco metropolitan area indicated that the overall economic impact of mixed CDW recycling is considerable without taking into account displaced industrial activities. When 650,000 tons of mixed CDW are recycled, the total impact on San Francisco metropolitan-wide employment and income is 440 employees and $23.122 million. However, if displaced industrial activities are taken into account, the total impact on metropolitan-wide employment and income declines to 106 employees and \$3.336million. The main cause of limited economic impact is that mixed CDW recycling is a highly mechanized process, and most employees consist of manual laborers. The potential impact on income and employment of mixed CDW recycling facilities does not significantly differ from that of the landfill industry. The next simulation showed that the extensive adoption of deconstruction would generate additional economic opportunities in the course of sustainable management of CDW. The net impact on employment and income in the San Francisco metropolitan area is 26 employees and \$0.635 million generated from 250 residential units of deconstruction.

The environmental impact of both mixed CDW recycling and deconstruction cases is modest regarding the heavy weight of CDW. Total reductions in energy use and GHG emissions are113 TJ and 10,701 tons from processing 650,000 tons of mixed CDW, and are 7.5 TJ and 605 TJ from deconstructing 250 housing units, respectively. One explanation for the moderate environmental benefits is that industrial activities assumed to be displaced in the simulation are not energy-intensive. In addition, off-site (transportation) environmental benefits from the CDW recycling and deconstruction are relatively small.

The simulation also examined the economic and environmental impact of the production of both recycled nylon 6 and recycled carpet padding in the Atlanta and Seattle metropolitan areas, respectively. The results of the simulation showed that both cases of waste carpet recycling would provide sizable economic opportunities to these metropolitan economies. From the production of 30 million pounds of recycled nylon 6, the impact on employment and income in the Atlanta metropolitan area would be 274 employees and $14.224 million, respectively, and from the production of 10 million pounds of recycled carpet padding, the impact on employment and income in the Seattle metropolitan area would be 43 employees and $1.798 million, respectively. However, when considering the displaced industrial activities in the rest of the nation, the net economic impact on the entire nation became significantly smaller in both cases. The positive economic impact is offset mostly by reductions in industrial activities pertaining to virgin nylon production and other manufacturing sectors.

In addition, the environmental benefits of carpet recycling differ according to applied techniques. The net environmental benefits derived from the production of recycled nylon 6 production are significant: Total net savings of energy and reductions in GHG emissions are 12,398 MJ per 1,000 pounds of recycled nylon 6 and 0.479 tons per 1,000 pounds, respectively. The environmental benefits derived from the production of recycled carpet padding are relatively smaller: Total net savings in energy and reductions in GHG emissions are 4,009 MJ per 1,000 pounds of recycled carpet padding and 0.282 tons per 1,000 pounds, respectively.

Regional environmental IO modeling simulations have shown that the establishment of recycling systems has a net positive, but relatively small impact on job creation, energy savings, and GHG emissions reduction in metro regions. This calls into question the accuracy and the margin of error of a regional environmental IO model for the recycling industry. The results should be carefully interpreted because they were a relatively small incremental change when considering as a proportion of the entire metropolitan economy or even as a proportion of the overall waste management industry.

Previous researchers have identified the causes of errors in the IO table and empirical analyses. One cause of errors is an inherent problem of using the survey method to measure economic transactions. Problems relating to inadequate survey design, classification, and definition, sampling, and poor training of respondents may contribute to errors in measurement that in turn lead to errors in the coefficients in the surveyed IO table (Jensen, 1980). The other type of errors may occur during the construction of the IO table. The accuracy of the IO tables may compromise during the reconciliation of the surveyed sales and purchase data and regionalization of technical coefficients (Stevens et al., 1988; Flegg et al., 1995; Brand, 1997).

To address the errors in the IO table, previous studies have examined stochastic errors and the variance of coefficients, and the confidence interval of multipliers (Quandt, 1958; West, 1986). In addition, Jackson (1986) provided insights into the statistical properties of the IO table. The probabilistic property of the IO table originated from not only errors in measurement and sampling, but also micro-level variations in production practices attributed to different location factors. Jackson suggested that understanding the distribution patterns of an individual firm allows interval estimation of output and multipliers instead of point estimates based on average industrial production practices. Given the technical coefficients distribution pattern of individual firms, a researcher can predict a probable range of outcomes in the absence of measurement and sampling errors. According to Jackson's simulations, a range for standard deviations of column multipliers was 0.0616 to 0.5880 while a range for mean of column multiplier distributions by a sectoral group was 1.4189 to 2.4221. In addition, variations of outputs were wider than those of multipliers.

Because this research utilizes the pre-existing regional IO table to which an econometric method that estimated the regional purchase coefficients (RPC) was applied, the regional environmental IO modeling of this research is subject to the same accuracy issues of the conventional regional IO table. Another type of error in this model relates to the distribution of industrial recycling activities. This research relies on a survey of a small number of carpet recycling companies in addition to secondary documents and personal contact with a particular company. If the observed industrial recycling activities deviate widely from average industrial recycling activities, the results of point estimates may be strongly biased. While this research conducts a sensitivity analysis for uncertain sources of information and important parameters, it does not fully examine the distribution of individual industrial recycling activities. Because this research has not analyzed enough samples to ascertain the probabilistic distribution of industrial recycling activities, a test of accuracy and a margin of error in the regional IO model for the recycling industry remains a future research topic. Jackson's research framework of a full probability distribution (1989) that links micro-level change with a macro-level structure may represent a promising direction for future research. As a growing number of companies are currently in the process of adopting a closed-loop production system and pursuing eco-efficiency strategies, an IO model that accounts for probable future change may prove useful as a tool for examining how micro-level changes toward closed-loop production systems will affect macro-level regional economic outcomes with interval estimates.

CHAPTER 8. IMPLICATIONS FOR SUSTAINABLE LOCAL ECONOMIC DEVELOPMENT

The purpose of this chapter is to reiterate the research questions and findings of theoretical and analytical models and to discuss the implications for sustainable local economic development. Recycling is a key dimension in urban economic and environmental sustainability. A thriving recycling system requires community-wide waste diversion, systematic institutional support, and a competent local and regional recycling industry. As urban and regional theory and planning research has largely disregarded the industrial respect of recycling, we have been left with little understanding of the industrial organization and spatial pattern of recycling.

Within this context, this dissertation explored two research questions that sought answers to1) what the logic of the industry organization and spatial pattern of recycling industry is and 2) to what extent recycling impacts the economy and the environment. With regard to the first question, the dissertation developed a theoretical model in light of institutional and organizational theory and presented two cases in different institutional contexts of local government responsibility and manufacturer responsibility. The theoretical model provided insights into how the recycling industry is organized and spatially distributed. The economic logic pertaining to the organization of the industry is essential to explaining the location of recycling facilities and their spatial linkages.

The theoretical model showed that local government responsibility represents a traditional localized recycling system in which a franchise waste management company plays an important role in investing in and operating the recycling infrastructure.

Recycling can become part of the consolidated waste management service of major waste management companies, which earn extra profits from waste that has already been collected. The case of CDW recycling in the San Francisco metropolitan area showed that local ordinances have prompted metropolitan-wide CDW recycling and that franchised waste management companies with local government policy support have played a key role in development of the collection and processing infrastructure. Economic benefits from the integration of multiple waste management services have dominated the location pattern of mixed CDW recycling facilities. The San Francisco metropolitan case indicates that mixed CDW recycling facilities, mainly operated by vertically-integrated waste management companies, co-located with other waste management facilities.

The manufacturer's responsibility for recycling is a relatively novel institutional approach. It created new dynamics to the existing landscape of the recycling industry. When manufacturers become involved in the management of their EOL products, recycling systems in terms of waste collection and a supply of recovered materials are not necessarily confined within a local area. A schematic framework conceptualized that the company's strategic decisions involving organizational forms, which are affected by feasible sets of recycling technology and existing industry structure, and the spatial patterns of recycling systems. The case of carpet recycling, built upon voluntary agreements for carpet stewardship, exemplified a diversified recycling system. Given the considerable autonomous role of industry, diverted waste carpet is recycled at in-house, outsourced, and independently-operated recycling facilities with different location patterns and service areas.

With regard to the second question, this dissertation constructed regional environmental IO models for three metropolitan areas to examine the economic and environmental impact of CDW and waste carpet recycling. Simulation of these models estimated their net economic and environmental impact by regarding both recycling and associated industrial activities and displaced industrial activities. The simulation results shed light on the following question: Is the recycling industry, as an industry target for sustainable local economic development, economically, socially, and environmentally justified? The quantitative results are interpreted from the perspective of the triple bottom line of sustainable local economic development: standard of living, equity, and the efficient use of resources.

The localized system of mixed CDW recycling resulted in increases in net employment and income in the San Francisco metropolitan area, but the magnitude of the net positive economic impact is moderate. Because a mixed CDW recycling facility is a highly mechanized process, it has limited job and income creation potential. In addition, the simulation showed that a wider application of the deconstruction technique added a small number of new job opportunities in the path of CDW recycling. Since many of the created jobs are low-skill and low-wage positions, CDW recycling can provide a local community with low-wage, unskilled job opportunities. With respect to its impact on the environment, mixed CDW recycling promoted efficient resource use in terms of recovering waste materials and energy savings as well as gave substantial benefits of saving landfill space although the energy savings per weight of CDW is relatively small.

Regarding the impact of carpet recycling, the simulation showed that waste carpet recycling with a regional-scale waste carpet collection system created the sizable net positive economic impact. The capital-intensive recycled nylon 6 production facility directly and indirectly provided hundreds of job opportunities to the metropolitan economy. However, the simulation showed that the impact of displaced jobs and income was also significant. The economic impact of the recycled nylon 6 production was substantially offset by that of displaced virgin material production activities. Thus, from the perspective of local economic development, averting the effect of local substitution may be a key element to retaining the positive economic impact on the locality. Finally, waste carpet recycling can reap a considerable environmental benefit. Despite the considerably longer distance required for transporting waste carpet, the production of recycled nylon 6 contributed to not only appreciable savings in energy but also a substantial reduction in GHG emissions.

Strategy of Planning for Sustainable Local Economic Development

From the findings of the research, this section suggests several implications for sustainable local economic development planning, including the cooperative effort of governments, financial support from a variety of sources, the determination of targeted waste material, market development for recycled materials, and mechanisms for enforcing policy.

Intergovernmental Cooperation: The research revealed that the diverse social regulatory schemes for recycling have been institutionalized. Typically, the promotion of recycling activities necessitates a formal or informal institutional rule that defines the covered material, collection and processing methods, and the responsible entity. Under the condition of political fragmentation, the inter-agency organization and the coordinated institutional regulations across local governments will facilitate the development of local or regional recycling systems. This research showed that the geographical area that a recycling facility serves ranged from at least multiple municipalities to multiple states. In the case of CDW recycling in the San Francisco metropolitan area, a single mixed CDW recycling facility could cover multiple cities and unincorporated counties. Therefore, cooperation among local governments may be a key factor that determines whether the operation of a recycling facility is cost effective and the implementation of the recycling program is successful.

Indeed, local governments in the San Francisco metropolitan area established a joint agency that dealt with solid waste management issues such as the Alameda County Waste Management Authority (ACWMA), comprised of 14 cities and two sanitary districts in Alameda County, and the South Bayside Waste Management Authority (SBWMA), comprised of 11 cities and one sanitary district in San Mateo County. The benefits of cooperation are that coordinated institutional regulations contribute to increasing the diverted volume of recyclable materials and providing a cost-effective recycling service. In the context of voluntary agreements, a coalition of environmental agencies of state governments could strengthen the negotiation power of governments when they set up an institution for recycling. Thus, a local economic development plan that targets recycling must incorporate multiple-jurisdictional contexts in which multiple local governments use various approaches to coordinate their programs as well as appropriate their spatial system boundaries of recycling.

Financial Support: Since the recycling process typically requires considerable up-front capital investment, the financial support may be an essential component of local policy. The carpet recycling survey indicated that the excessive up-front capital investment poses one of the greatest challenges to establishing a recycling infrastructure. This research illustrated that in one case, one state government revolving loan program helped a small business, LA Fiber, launch a local recycling facility by modifying existing processing equipment, and in another case, a local government grant helped Waste Management, Inc., a franchise waste management company, to expand its recycling capacity.

Performance-based financial support, which is financial support proportional to the amount of waste a recycling facility processes, is another useful policy tool. The Disposal Facility Tax in the city of San Jose and ACWMA assistance to a mixed CDW recycling facility was an exemplary case of performance-based financial support. Such support may temporarily protect a recycling business from the price volatility of recovered materials.

Targeted Waste Materials: The simulation shows that the economic value and the environmental benefits of recycling can widely differ depending on the type of material being recycled. Even if the same material is being recycled, the application of the various recycling techniques may result in different economic and environmental effects such as those associated with waste carpet recycling. For the appropriate selection of waste materials, a local economic development plan should consider multiple aspects including the potential volume of divertible waste materials, market conditions for recycled material, the economic impact potential and environmental benefits of recycling, and the structures of existing industries. A particularly important step is the identification of existing local businesses that could potentially process diverted waste materials and consume recovered materials. Thus, a strategy for attracting recycling business should be developed.

Market Development for Recycled Materials: Market development for recycled materials is a key dimension of the promotion of the recycling industry. The case in which recycling becomes part of a vertical integrated production system may prove an ideal model of the stable internal consumption of recovered materials. If it does not, recycled material must compete with virgin material in the market. A market development plan can foster the expansion of the recycled materials market by promoting wider adoption of a green procurement policy, green supply chains, green building certification, and green building code, as well as providing informational services such as a recycled product and recycling company directory.

A useful policy for promoting recycled material market, one that a local government has direct control over is the green building code (Cicro, 2007). Green building codes have widely been adopted in the many city and county governments. Green building codes of the local governments in California commonly utilize the rating systems such as the Leadership in Energy and Environmental Design (LEED) developed by the United States Green Building Council (USGBC) and the GreenPointRated developed by the Build It Green. The building construction or alteration projects are required to meet the specified standard or to earn a certain level of points in the LEED or GreenPointRated. Those rating systems have categories of material and resource use. When builders uses the recycled-content or salvaged materials at a certain percentage, they can earn some points. Although green building codes do not directly mandate the use of recycled or salvage materials, builders can be incentivized to use the recycled-content building materials through the rating system. The use of the rating systems in green building codes is currently a major local policy tool for promoting the demand of recycled products.

Mechanisms for Enforcing Policy: The mechanism for enforcing policy may contribute to ensuring the fulfillment of a voluntary agreement or a recycling program. Cases of voluntary agreements on waste carpet stewardship have emphasized the necessity of a policing mechanism. A local recycling program with such an enforcement policy can be a foundation upon which a local recycling business is launched and continues to operate. An example of a strong, enforceable mechanism is the deposit program of the construction and demolition permit.

Limitations and Future Research

The theoretical and analytical models presented in this dissertation have several limitations. First, the research presented a theoretical model explaining the logic of industry organizations and spatial patterns of recycling and showed illustrative cases representing proposed patterns; the number of cases of recycling facilities, however, was not sufficient for statistical inference. In particular, EPR policy is a relatively new phenomenon. In the case of carpet recycling, even though voluntary carpet recycling efforts were made over a ten-year time span, the carpet recycling industry is still immature. Hence, future research could expand the scope of geographical areas for statistical analysis, and once EPR policy has been more widely adopted, it could revisit this topic in a more extensive investigation of recycling systems.

The work of regional environmental IO models confronted several challenging issues. Building a regional environmental inventory suffered from insufficient sub-national energy use statistics by industry sector. While the intention of the regional environmental IO model was to build a metropolitan-specific model, the research relied primarily on state- or census region-level data. The wider and inconsistent spatial units for regionalizing energy use and GHG emission coefficients may have diminished the precision of the impact analysis. In addition, the criteria pollutant and toxic release are not modeled in the analysis. Those could be added to a regional environmental inventory connected to IO tables because the National Emission Inventory and Toxic Release Inventory contain specific location information. The regionalization method for energy use coefficients and the expansion of a regional environmental inventory also call for further investigation.

Occasionally, since environmental studies investigated detailed production process or a specific product, the addition or disaggregation of an industry sector in the IO model is necessary. It requires extensive research efforts for the compilation of relevant data. In the absence of a relevant comprehensive source of data, the research should rely on a variety of sources such as business surveys, site visits, engineering models, and secondary documents, which may incur a problem about a data compilation process. The ad hoc data compilation process would compromise the reliability of the data. For example, when data are obtained from different sources, they can be inconsistent, or if obtained from a single business case, they can be biased. Thus, to improve the reliability of analysis, future research should pay attention to develop and then follow a systematic process of compiling data.

Finally, this research established two-region environmental IO models for three metropolitan cases. Conceptually, this IO modeling framework can be extended to an actual multi-region IO model. Thus, an ambitious follow-up study would entail the construction of a multi-regional environmental IO model for all U.S. states.

APPENDIX

Carpet Recycling Survey

The carpet recycling survey was conducted for understanding the current status of carpet recycling industrial activities as well as providing the economic information such as cost and revenue for building a regional environmental IO model. The carpet recycling survey forms were designed for four types of carpet recycling companies: collection, sorting, processing, and end-use. The contact information of carpet recycling companies was compiled from the website of the CARE and with the cooperation of the Seattle Public Utilities: 104 carpet recycling companies and associated contact information were identified. The online survey method was employed, and the survey form was distributed three times from July 7th to October 11th, 2011. 36 companies responded to the carpet recycling survey, and the response rate was 35%. After the completion of carpet survey, the survey results and major findings were distributed to carpet recycling companies and the director of the CARE. Followings are the major findings that sent to the carpet recycling companies.

Collection: The collection of waste carpet is carried out at a relatively small scale. The average amount of waste carpet collected by each company is less than 1,000 tons annually. However, a major carpet collecting company collected thousands of tons of waste carpet. Respondents were asked to identify their collection costs. For most, collection costs were less than $0.05 per pound. The respondents charged a fee to either those who discarded waste carpet or to sorting and processing companies to collect the used carpet. The average range of the collection fee was $0.05 -$0.06 per pound.

That waste carpet collection is often a small-scale operation, or waste carpet is a small portion of other types of waste collected by a larger waste collection company, makes it difficult to determine number of employees and occupations directly involved in the waste carpet collection. Employees are typically hired for a multi-task position that includes driving, equipment operation, and manual work. A small collector is typically a sole proprietorship that hires manual workers temporarily as needed. The respondents to our survey indicated the typical small carpet collection firm hires three to five employees per year. The average hourly wage for driver, equipment operator, and manual worker ranges from $10 - $12.

Sorting: The typical waste carpet sorting firm is small, hires on average 10 employees, and sorts thousands of tons of waste carpet per year. Half of the sorting companies who responded are also directly involved in the collection of waste carpet. The sorting companies hired manual workers (57%), drivers (15%), and equipment operators (13%). The average hourly wage of these occupations ranged from $10 - $15. Sorted materials consisted of Nylon 6 (35%), Nylon 6.6 (26%), Polypropylene (11%), and PET (27%). The average sorting cost is less than $0.05 per pound. Sorting companies generated revenue through fees collected from those who discard waste carpet or through selling sorted materials to processing companies. While waste carpet is typically collected within the metropolitan area or state, sorted waste carpet materials are delivered to multiple states.

Processing and end use: The typical processing company operates at a larger scale than a collection or sorting company. However, there is variation in the size of processing companies. While some small-scale processors handle only hundreds of tons of waste carpet, larger ones process multiple thousands of tons of waste carpet. Nylon 6 and Nylon 6.6 make up more than three quarters of processed waste carpeting materials. The most common recycled material that processing companies manufactured is plastic pellets. Recycled carpet face fiber, calcium carbonate, and mixed fiber from backing are other types of recycled materials produced. Recycled face fiber is the most valuable output. Processing cost mostly ranged from $0.31 to $0.60. Occupations of employees are relatively diverse in the processing companies: equipment operators (37%), manual labor (30%), driver (12%), managerial (10%), administration (4%), sales (2%), and research (2%). The larger scale of processing operation is a more capital intensive process. The processing companies serve a wider scale of geographic areas, at least multiple states and the U.S.

Opinion: Our survey examined opinions on the key barriers and challenges to growing the carpet recycling industry. The most serious challenges (more than 80% agreement) identified by respondents were the volatile market price of recycled materials and excessive up-front capital investments. An uncertain supply of waste materials, technical difficulties, and the current economic recession were the second largest barriers (60% - 75% agreement). Processing companies were concerned about the lack of collection infrastructure. Of least concern was the lack of skilled labor (20% agreement). The survey also investigated opinions about the usefulness of policy supports for the carpet recycling industry. Overall, the respondents supported all suggested policy options, but rated financial support, information services, and market development as the most important (75%-79% useful).The respondents ranked regulation options, such as a landfill ban or a mandatory recycling local ordinance, as less useful than other policy supports (60%-70% useful).

REFERENCES

Albino, V., & Kühtz, S. (2004). Enterprise input-output model for local sustainable development--the case of a tiles manufacturer in Italy. *Resources, Conservation and Recycling, 41*(3), 165-176.

Ayres, R. U., & Kneese, A. V. (1969). Production, consumption, and externalities. *American Economic Review, 59*(3), 282.

Binder, M., Albrecht, S., Marincovic, C., Baer, S., McGavis, D., & Harless, D. (2010). *Life Cycle Assessment of Caprolactam Production from Nylon 6 Carpet Recycling*. Poster presented at the Life Cycle Assessment X: Bridging Science, Policy, and the Public, Portland, Oregon.

Brand, S. (1997). On the appropriate use of location quotients in generating regional input-output tables: A comment. *Regional Studies, 31*(8), 791.

Briassoulis, H. (1986). Integrated economic-environmental-policy modeling at the regional and multiregional level: Methodological characteristics and issues. *Growth & Change, 17*(3), 22.

Brown, M. A., Southworth, F., & Sarzynski, A. (2008). Shrinking the carbon footprint of metropolitan America. Washington, D.C.: The Brookings Institution

California Integrated Waste Management Board. (2003). *The Capitol Area East End Office Complex: A Case for Construction and Demolition Waste Diversion*. Sacramento, CA: Author. Retrieved from www.calrecycle.ca.gov/publications/GreenBuilding/43303023.doc

California Integrated Waste Management Board. (2006). *Detailed Characterization of Construction and Demolition Waste.* Sacramento, CA: Cascadia Consulting Group. Retrieved from http://www.calrecycle.ca.gov/publications/Disposal/34106007.pdf

Carpet America Recovery Effort (2010). *CARE 2010 annual report*: Carpet America Recovery Effort. Dalton, Georgia: Author

Chertow, M. R. (2007). "Uncovering" industrial symbiosis. *Journal of Industrial Ecology, 11*(1), 11-30.

Choi, T., Jackson, R. W., Leigh, N. G., & Jensen, C. D. (2011). A baseline input—output model with environmental accounts (IO_{EA}) applied to e-waste recycling. *International Regional Science Review, 34*(1), 3-33.

Christoff, P. (1996). Ecological modernisation, ecological modernities. *Environmental Politics, 5*(3), 476.

Cicas, G., Hendrickson, C., Horvath, A., & Matthews, H. (2007). A regional version of a US economic input-output life-cycle assessment model. *The International Journal of Life Cycle Assessment, 12*(6), 365-372.

Circo, C. (2008). Using mandates and incentives to promote sustainable construction and green building projects in the private sector: A call for more state land use policy initiatives. *Penn State Law Review, 112*, 731-782.

City of Portland, Portland Development Commission. (2009). *Economic Development Strategy: A Five-Year Plan for Promoting Job Creation and Economic Growth*. Portland, Oregon: Author. Retrieved from http://pdxeconomicdevelopment.com/docs/Portland-Ec-Dev-Strategy.pdf

City of San Jose. (2008). *City of San Jose C&D Characterization Study*. California: Cascadia Consulting Group, Inc. Retrieved from http://www.sjrecycles.org/construction-demolition/pdf/cddd_WasteCharacterizationReport_11-08.pdf

Clinton, J. A. (1999). Putting industrial ecology into place: Evolving roles for planners. *Journal of the American Planning Association, 65*(4), 364.

Coase, R. H. (1937). The nature of the firm. *Economica, 4*(16), 386-405.

Cochran, K., Townsend, T., Reinhart, D., & Heck, H. (2007). Estimation of regional building-related C&D debris generation and composition: Case study for Florida, US. *Waste Management, 27*(7), 921-931.

Cohen, M. J. (1997). Risk society and ecological modernisation alternative visions for post-industrial nations. *Futures, 29*(2), 105-119.

Davis, C. S., Diegel, W. S., Boundy. R. G. (2008). *Transportation energy data book: Edition 27* (ORNL-6981). Washington, D.C.: U.S. Department of Energy. Retrieved from http://www.cleanenergycouncil.org/files/Edition27_Full_Doc.pdf

Desrochers, P. (2001). Cities and industrial symbiosis: some historical perspectives and policy implications. *Journal of Industrial Ecology, 5*(4), 29-44.

Deutz, P., & Gibbs, D. (2004). Eco-industrial development and economic development: industrial ecology or place promotion? *Business Strategy & the Environment, 13*(5), 347-362.

Dodman, D. (2009). Blaming cities for climate change? An analysis of urban greenhouse gas emissions inventories. *Environment and Urbanization, 21*(1), 185-201.

Duchin, F. (2004). *Input-output economics and material flows* (Working Paper No. 0424). Retrieved from Rensselaer Polytechnic Institute Website: http://www.economics.rpi.edu/workingpapers/rpi0424.pdf

Dunn, B. C., & Steinemann, A. (1998). Industrial ecology for sustainable communities. *Journal of Environmental Planning & Management, 41*(6), 661.

Esty, D. C., & Porter, M. E. (1998). Industrial ecology and competitiveness. *Journal of Industrial Ecology, 2*(1), 35-43.

Fishbein, B. K. (2000). Carpet take-back: EPR American style. *Environmental Quality Management, 10*(1), 25-36.

Fisher, D. R., & Freudenburg, W. R. (2001). Ecological modernization and its critics: Assessing the past and looking toward the future. *Society & Natural Resources, 14*(8), 701-709.

Fitzgerald, J. (2010). *Emerald cities: urban sustainability and economic development*. New York: Oxford University Press.

Fitzgerald, J., & Leigh, N. G. (2002). *Economic revitalization: Cases and strategies for city and suburb*. Thousand Oaks, Calif.: Sage Publications.

Flegg, A. T., Webber, C. D., & Elliott, M. V. (1995). On the appropriate use of location quotients in generating regional input–output tables. *Regional Studies, 29*(6), 547-561.

Fuellhart, K. (1999). Localization and the use of information sources. *European Urban and Regional Studies, 6*(1), 39-58.

Gao, W., Ariyama, T., Ojima, T., & Meier, A. (2001). Energy impacts of recycling disassembly material in residential buildings. *Energy and Buildings, 33*(6), 553-562.

Georgia Department of Natural Resources (2010). *Statewide construction and demolition debris characterization study.* Atlanta, Georgia: R.W. Beck. Retrieved from http://www.gasustainability.org/sites/uploads/sustain/pdf/FinalCDReport.pdf

Gibbs, D. (2000). Ecological modernisation, regional economic development and regional development agencies. *Geoforum, 31*(1), 9-19.

Gibbs, D. (2006). Prospects for an environmental economic geography: Linking ecological modernization and regulationist Approaches. *Economic Geography, 82*(2), 193.

Gibbs, D., Deutz, P., & Proctor, A. M. Y. (2005). Industrial ecology and eco-industrial development: A potential paradigm for local and regional development? *Regional Studies, 39*(2), 171-183.

Guidry, C. (2008). *Modified comparative life cycle assessment of end-of-life options for post-consumer products in urban regions* (Master's Thesis). Georgia Institute of Technology, Atlanta, Georgia.

Guy, B. (2001). *Building deconstruction: reuse and recycling of building materials*. Gainesville, Florida: Center for Construction and Environment. Retrieved from http://www.dep.state.fl.us/waste/quick_topics/publications/shw/recycling/InnovativeGrants/IGyear2/reports/alachua2.pdf

Haberl, H., Fischer-Kowalski, M., Krausmann, F., Weisz, H., & Winiwarter, V. (2004). Progress towards sustainability? What the conceptual framework of material and energy flow accounting (MEFA) can offer. *Land Use Policy, 21*(3), 199-213.

Hajer, M. A. (1995). *The politics of environmental discourse: ecological modernization and the policy process*. New York: Clarendon Press.

Haughton, G., & Counsell, D. (2004). *Regions, spatial strategies, and sustainable development*. London; New York: Routledge.

Heeres, R. R., Vermeulen, W. J. V., & de Walle, F. B. (2004). Eco-industrial park initiatives in the USA and the Netherlands: first lessons. *Journal of Cleaner Production, 12*(8-10), 985-995.

Hendrickson, C. T., Horvath, A., Joshi, S., Klausner, M., Lave, L. B., & McMichael, F. C. (1997). *Comparing two life cycle assessment approaches: a process model vs. economic input-output-based assessment.* Proceedings of the IEEE International Symposium on Electronics and the Environment. San Francisco, California. Retrieved from http://ieeexplore.ieee.org/stamp/stamp.jsp?arnumber=00605313

Hendrickson, C. T. (2006). *Environmental life cycle assessment using economic input-output analysis*. Washington, D.C.: Resources for the Future.

Hillman, T., & Ramaswami, A. (2010). Greenhouse gas emission footprints and energy use benchmarks for eight U.S. cities. *Environmental Science & Technology, 44*(6), 1902-1910.

Hoekstra, R., & van den Bergh, J. C. J. M. (2006). Constructing physical input-output tables for environmental modeling and accounting: Framework and illustrations. *Ecological Economics, 59*(3), 375-393.

Hoornweg, D., Sugar, L., & Trejos Gómez, C. L. (2011). Cities and greenhouse gas emissions: moving forward. *Environment and Urbanization, 23*(1), 207-227.

Horvath, A. (2004). Construction materials and the environment. *Annual Review of Environment & Resources, 29*(1), 181.

Huang, G. H., Anderson, W. P., & Baetz, B. W. (1994). Environmental input-output analysis and its application to regional solid-waste management planning. *Journal of Environmental Management, 42*(1), 63-79.

Huang, S.-L., & Hsu, W.-L. (2003). Materials flow analysis and emergy evaluation of Taipei's urban construction. *Landscape and Urban Planning, 63*(2), 61-74.

International City/County Management Association. (2007), *ICMA Profile of Local Government Service Delivery Choices 2007.* Washington D.C.: Author http://icma.org/en/icma/knowledge_network/documents/kn/Document/100022/ICMA_Profile_of_Local_Government_Service_Delivery_Choices_2007

Isard, W. (1968) On the linkage of socio-economic and ecological systems. *Papers Regional Science Association* 21, 79-99.

Jackson, R. W. (1986). The full-distribution approach to aggregate representation in the input-output modeling framework. *Journal of regional science,* 26**,** 515-531.

Jackson, R. W. (1989). Probabilistic input-output analysis: Modeling directions. *Socio-economic planning sciences,* 23**,** 87-95.

Jackson, R. W. (1998). Regionalizing national commodity-by-industry accounts. *Economic Systems Research, 10*(3), 223.

Jackson, R. W., Choi, T., & Leigh, N. G. (2008). *Recycling and remanufacturing in input-output models* (Research Paper 2008-4). Morgantown, West Virginia: Regional Research Institute.

Jackson, R. W., & Schwarm, W. R. (2011). Accounting foundations for interregional commodity-by-industry input-output models. *Letters in Spatial and Resource Sciences*, 4, 187-196.

Jänicke, M. (2008). Ecological modernisation: new perspectives. *Journal of Cleaner Production, 16*(5), 557-565.

Jepson, E. J. (2004). The adoption of sustainable development policies and techniques in U.S. cities. *Journal of Planning Education and Research, 23*(3), 229-241.

Jensen, R. C. (1980). The concept of accuracy in regional input-output models. *International regional science review, 5*(2), 139.

Johnson, M. H., & Bennett, J. T. (1981). Regional environmental and economic impact evaluation: An input-output approach. *Regional Science and Urban Economics, 11*(2), 215-230.

Joshi, S. (1999). Product environmental life-cycle assessment using input-output techniques. *Journal of Industrial Ecology, 3*(2/3), 95-120.

Joskow, P. L. (1988). Asset specificity and the structure of vertical relationships: Empirical evidence. *Journal of Law, Economics, & Organization, 4*(1), 95-117.

Kelly, T. (1998). *Crushed cement concrete substitution for construction aggregates: A materials flow analysis*. (U.S. Geological Survey Circular 1176). U.S. Department of Interior.

Kennedy, C., Ramaswami, A., Dhakal, S., & Carney, S. (2009). *Greenhouse gas emission baselines for global cities and metropolitan regions.* Paper presented at the 5th Urban Research Symposium: Cities and Climate Change - Responding to an Urgent Agenda, Marseille, France. Retrieved from http://siteresources.worldbank.org/INTURBANDEVELOPMENT/Resources/336387-1256566800920/6505269-1268260567624/KennedyComm.pdf

King County Department of Natural Resources and Parks. (2006). *Waste Monitoring Program: Market Assessment for Recyclable Materials in King County*. Washington: Cascadia Consulting Group, Inc. Retrieved from http://your.kingcounty.gov/solidwaste/about/documents/MarketsReportFINAL.pdf

King County Department of Natural Resources and Parks. (2009). *2007/2008 Construction and Demolition Materials Characterization Study*. Washington: Cascadia Consulting Group, Inc. Retrieved from http://your.kingcounty.gov/solidwaste/about/documents/CD-characterization-study-2008.pdf

Lange, G.-M. (1998). Applying an integrated natural resource accounts and input-output model to development planning in Indonesia. *Economic Systems Research, 10*(2), 113.

Lave, L., Conway-Schempf, N., Harvey, J., Hart, D., Bee, T., & MacCracken, C. (1998). Recycling postconsumer nylon carpet. *Journal of Industrial Ecology, 2*(1), 117-126.

Leigh, N. G., & Patterson, L. M. (2006). Deconstructing to redevelop: A sustainable alternative to mechanical demolition. *Journal of the American Planning Association, 72*(2), 217-225.

Leigh, N. G., Choi, T., & Hoelzel, Z. N. (2012) New Insights into Electronic Waste Recycling in Metropolitan Areas. Manuscript accepted in *Journal of Industrial Ecology*

Leigh, N. G., Realff, M. J., Ai, N., French, S. P., Ross, C. L., & Bras, B. (2007). Modeling obsolete computer stock under regional data constraints: An Atlanta case study. *Resources, Conservation and Recycling, 51*(4), 847-869.

Lenzen, M. (2000). Errors in conventional and input-output—based life—cycle inventories. *Journal of Industrial Ecology, 4*(4), 127-148.

Lenzen, M., Pade, L.-L., & Munksgaard, J. (2004). CO_2 multipliers in multi-region input-output models. *Economic Systems Research, 16*(4), 391-412.

Lenzen, M., & Peters, G. M. (2010). How city dwellers affect their resource hinterland. *Journal of Industrial Ecology, 14*(1), 73-90.

Leontief, W. (1970). Environmental repercussions and the economic structure: An input-output approach. *The Review of Economics and Statistics, 52*(3), 262-271.

Leroux, K., & Seldman, N. (2000). *Deconstruction: salvaging yesterday's buildings for tomorrow's sustainable communities*. Washington D.C: Institute for Local Self-Reliance Retrieved from http://www.ilsr.org/recycling/decon/Deconstruction.pdf

Levis, J. W. (2008). *A life-cycle analysis of alternatives for the management of waste hot-mix asphalt, commercial food waste, and construction and demolition waste.* (Master's Thesis), North Carolina State University, Raleigh, North Carolina.

Lifset, R. (1993). Take it back: Extended producer responsibility as a form of incentive-based environmental policy. *Journal of Resource Management and Technology, 21*(4), 163-175.

Lifset, R. (2009). Industrial ecology in the age of input-Output analysis. In S. Suh (Ed.), *Handbook of Input-Output Economics in Industrial Ecology* (pp. 263-284). New York: Springer.

Lin, X., & Polenske, K. R. (1998). Input--output modeling of production processes for business management. *Structural Change and Economic Dynamics, 9*(2), 205-226.

Lindhqvist, T., & Lifset, R. (2003). Can we take the concept of individual producer responsibility from theory to practice? *Journal of Industrial Ecology, 7*(2), 3-6.

Little Hoover Commission. (1989). *Report on Solid Waste Management: The Trashing of California*. Sacramento, California: Author. Retrieved from http://www.lhc.ca.gov/studies/096/report96.PDF.

Lounsbury, M., Ventresca, M., & Hirsch, P. M. (2003). Social movements, field frames and industry emergence: a cultural–political perspective on US recycling. *Socio-Economic Review, 1*(1), 71-104.

Louis, G. E. (2004). A historical context of municipal solid waste management in the United States. *Waste Management & Research, 22*(4), 306-322.

Lyon, T. P., & Maxwell, J. W. (2008). Corporate social responsibility and the environment: A theoretical perspective. *Review of environmental economics and policy 2*(2), 240.

Lyons, D. I. (2007). A spatial analysis of loop closing among recycling, remanufacturing, and waste treatment firms in Texas. *Journal of Industrial Ecology, 11*(1), 43-54.

Marinkovic, S., Radonjanin, V., Malesev, M., & Ignjatovic, I. (2010). Comparative environmental assessment of natural and recycled aggregate concrete. *Waste Management, 30*(11), 2255-2264.

Massachusetts Department of Environmental Protection. (2008). *2007 Massachusetts Construction and Demolition Debris Industry Study.* Massachusetts: DSM Environmental Service, Inc. Retrieved from http://www.mass.gov/dep/recycle/reduce/07cdstdy.pdf

Miller, R. E., & Blair, P. D. (2009). *Input-output analysis: foundations and extensions* (2nd ed.). Cambridge England; New York: Cambridge University Press.

Mol, A. P. J. (1997). Ecological modernization: Industrial transformation and environmental reform. In M. R. Redclift & G. Woodgate (Eds.), *The international handbook of environmental sociology* (pp.138-149). Cheltenham, UK; Northampton, MA, USA: Edward Elgar.

Mol, A. P. J. (1999). Ecological modernization and the environmental transition of Europe: between national variations and common denominators. *Journal of Environmental Policy & Planning, 1*(2), 167-181.

Munday, M., & Roberts, A. (2006). Developing approaches to measuring and monitoring sustainable development in Wales: A review. *Regional Studies, 40*(5), 535-554.

Munksgaard, J., Wier, M., Lenzen, M., & Dey, C. (2005). Using input-output analysis to measure the environmental pressure of consumption at different spatial levels. *Journal of Industrial Ecology, 9*(1-2), 169-185.

Nakamura, S., & Kondo, Y. (2002). Input-Output Analysis of Waste Management. *Journal of Industrial Ecology, 6*(1), 39-63.

Nakamura, S., & Kondo, Y. (2006a). A waste input-output life-cycle cost analysis of the recycling of end-of-life electrical home appliances. *Ecological Economics, 57*(3), 494-506.

Nakamura, S., & Kondo, Y. (2006b). Hybrid LCC of appliances with different energy efficiency. *The International Journal of Life Cycle Assessment, 11*(5), 305 - 314.

Nakamura, S., Nakajima, K., Yoshizawa, Y., Matsubae-Yokoyama, K., & Nagasaka, T. (2009). Analyzing polyvinyl chloride in Japan with the waste input–output material flow analysis model. *Journal of Industrial Ecology, 13*(5), 706-717.

North Central Texas Council of Governments (2007). *Construction and demolition material recovery facility feasibility study.* Texas: R. W. Beck. Retrieved from http://www.nctcog.org/envir/SEELT/reduction/RWBeckCDMRFFeasibilityStudy_Final.pdf

Orange County, Seminole County. (2003). *Innovative drywall recycling grant*. Florida: R.W. Beck. Retrieved from http://www.dep.state.fl.us/waste/quick_topics/publications/shw/recycling/InnovativeGrants/IGyear3/finalreports/OrangeFinalRpt.pdf

Pagell, M., Wu, Z., & Murthy, N. N. (2007). The supply chain implications of recycling. *Business Horizons, 50*(2), 133-143.

Patterson, L. M. (2007). *Local economic development agencies' support for construction & demolition recycling*. (Doctoral Dissertation), Georgia Institute of Technology, Atlanta Georgia.

Pedersen, O. G. d., & Haan, M. (2006). The system of environmental and economic accounts-2003 and the economic relevance of physical flow accounting. *Journal of Industrial Ecology, 10*(1-2), 19-42.

Pellow, D. N., Schnaiberg, A., & Weinberg, A. S. 1999. *Putting the ecological modernization thesis to the test: The promises and performance of urban recycling*. Retrieved from http://www.ipr.northwestern.edu/publications/workingpapers/2004/schnaiberg/13aTestEcolMod.pdf

Pimenteira, C. A. P., Carpio, L. G. T., Rosa, L. P., & Tolmansquim, M. T. (2005). Solid wastes integrated management in Rio de Janeiro: input-output analysis. *Waste Management, 25*(5), 539-553.

Quandt, R. E. (1958) Probabalistic errors in the Leontief system. *Naval Research Logistic Quarterly, 5***,** 155-170.

Ramaswami, A., Hillman, T., Janson, B., Reiner, M., & Thomas, G. (2008). A demand-centered, hybrid life-cycle methodology for city-scale greenhouse gas inventories. *Environmental Science & Technology, 42*(17), 6455-6461.

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.P, Suh, S., Weidema, B.P., and Pennington, D. W. (2004). Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications. *Environment International, 30*(5), 701-720

RCF Economic & Financial Consulting, Chicago Metropolitan Agency for Planning. (2009). *Green Economic Development Strategies for the Chicago Region*. Chicago, Illinois: Delta Redevelopment Institute. Retrieved from http://www.delta-institute.org/publications/DeltaREDI_CMAP_GreenEconDevtReport_June2009_v2.pdf

Reiff, T. (2010). *From Solid Waste to Economic Development through Deconstruction, Distribution and Reuse* [PowerPoint slides]. Atlanta, Georgia.

Reinhardt, F. (1999). Market failure and the environmental policies of firms: economic rationales for "beyond compliance" behavior. *Journal of Industrial Ecology, 3*(1), 9-21.

Robinson, G. R., Menzie, W. D., & Hyun, H. (2004). Recycling of construction debris as aggregate in the Mid-Atlantic Region, USA. *Resources, Conservation and Recycling, 42*(3), 275-294.

Rosenblum, J., Horvath, A., & Hendrickson, C. (2000). Environmental implications of service industries. *Environmental Science & Technology, 34*(22), 4669-4676.

Saha, D., & Paterson, R. G. (2008). Local government efforts to promote the “three Es” of sustainable development. *Journal of Planning Education and Research, 28*(1), 21-37.

Sahely, H. R., Dudding, S., & Kennedy, C. A. (2003). Estimating the urban metabolism of Canadian cities: Greater Toronto Area case study. *Canadian Journal of Civil Engineering, 30*(2), 468.

San Francisco Planning Department. (2011). *San Francisco Housing Inventory 2010*. California: Author. Retrieved from http://www.sf-planning.org/ftp/files/publications_reports/2010_Housing_Inventory_Report.pdf

Schlosberg, D., & Rinfret, S. (2008). Ecological modernisation, American style. *Environmental Politics, 17*(2), 254-275.

Scott, A. J. (1983). Industrial organization and the logic of intra-metropolitan location: I. theoretical considerations. *Economic Geography, 59*(3), 233-250.

Scott, A. J. (1986). Industrial organization and location: division of labor, the firm, and spatial process. *Economic Geography, 62*(3), 215-231.

Segars, J. W., Bradfield, S. L., Wright, J. J., & Realff, M. J. 2003. EcoWorx, green engineering principles in practice. *Environmental Science & Technology, 37*(23), 5269-5277.

Shaw Industries Group, Inc. (2009). Corporate Sustainability Report 2009. Dalton, Georgia: Author. Retrieved from http://www.shawgreenedge.com/Shaw_Sustainabilty_Report_2009.pdf

Sonnenfeld, D. A. (2002). Social movements and ecological modernization: The transformation of pulp and paper manufacturing. *Development & Change, 33*(1), 1.

Steenge, A. E. (1999). Input-output theory and institutional aspects of environmental policy. *Structural Change and Economic Dynamics, 10*(1), 161-176.

Sterr, T., & Ott, T. The industrial region as a promising unit for eco-industrial development--reflections, practical experience and establishment of innovative instruments to support industrial ecology. *Journal of Cleaner Production, 12*(8-10), 947-965.

Stevens, B. H., Treyz, G. I., & Lahr, M. L. (1988). *On the Comparative Accuracy of RPC Estimating Techniques*. Regional Science Research Institute.

Subbiah, V. (2008). *Sustainability studies in recycling post consumer carpet* (Master's Thesis). Georgia Institute of Technology, Atlanta, Georgia.

Suh, S. (2004). Functions, commodities and environmental impacts in an ecological-economic model. *Ecological Economics, 48*(4), 451-467.

Suh, S., & Huppes, G. (2009). Methods in the life cycle inventory of a product. In S. Suh (Ed.), *Handbook of Input-Output Economics in Industrial Ecology* (pp. 263-284). New York: Springer.

Suh, S., & Kagawa, S. (2005). Industrial ecology and input-output economics: an introduction. *Economic Systems Research, 17*(4), 349-364.

Suh, S., Lenzen, M., Treloar, G. J., Hondo, H., Horvath, A., Huppes, G., Jolliet, O., Klann, U., Krewitt, W., Moriguchi, Y., Munksgaard, J., & Norris, G. (2003). System boundary selection in life-cycle inventories using hybrid approaches. *Environmental Science & Technology, 38*(3), 657-664.

Suh, S., B. Weidema, J. H. Schmidt, and R. Heijungs. (2010). Generalized make and use framework for allocation in life cycle assessment. *Journal of Industrial Ecology,* 14(2), 335-353.

Tam, V. W. Y., & Tam, C. M. (2006). A review on the viable technology for construction waste recycling. *Resources, Conservation and Recycling, 47*(3), 209-221.

Toffel, M. W. (2003). The growing strategic importance of end-of-life product management. *California Management Review, 45*(3), 102.

Toffel, M. W., Stein, A., & Lee, K. L. (2008). *Extending producer responsibility: An evaluation framework for product take-back policies* (Working paper no. 09-026). Retrieved from Harvard Business School website: http://www.hbs.edu/research/pdf/09-026.pdf

Toronto Economic Development, Culture and Tourism (2007). *People, planet & profit: catalyzing economic growth & environmental quality in the city of Toronto.* Toronto, Canada: Delphi Group, Gartner Lee Limited. Retrieved from http://www.toronto.ca/business_publications/pdf/green_economic_development_22may2007.pdf

Turner, K. (2006). Additional precision provided by region-specific data: The identification of fuel-use and pollution-generation coefficients in the Jersey economy. *Regional Studies, 40*(4), 347-364.

U.S. Census Bureau (2004). *2002 Vehicle Inventory and Use Survey.* [Data file]. Retrieved from http://www.census.gov/svsd/www/vius/2002.html

U.S. Census Bureau (1997). *1997 Economic Census.* [Data file]. Retrieved from http://www.census.gov/epcd/www/econ97.html

U.S. Census Bureau (2007). *2007 Economic Census.* [Data file]. Retrieved from http://www.census.gov/econ/census07/

U.S. Department of Agriculture. (2009). *2007 Census of Agriculture* (AC-07-A-51). Washington, D.C.: Author. Retrieved from http://www.agcensus.usda.gov/Publications/2007/Full_Report/usv1.pdf

U.S. Department of Transportation. (2010). 2007 Commodity Flow Survey [Data file], Retrieved from http://www.bts.gov/publications/commodity_flow_survey/

U.S. Department of Transportation, Research and Innovative Technology Administration, Bureau of Transportation Statistics. (2011). *Transportation satellite accounts: A look at transportation's role in the economy,* Washington, D. C.: Author. 2011. Retrieved from http://www.bts.gov/publications/transportation_satellite_accounts/2011/pdf/entire.pdf

U.S. Energy Information Administration. (2006). *Manufacturing Energy Consumption Survey* [Date File]. Retrieved from: http://205.254.135.7/emeu/mecs/mecs2006/2006tables.html

U.S. Energy Information Administration. (2007). *Annual energy review 2006* (DOE/EIA-0384). Washington, D.C.: Author. Retrieved from http://www.me.mtu.edu/~jstallen/courses/MEEM4200/lectures/energy_intro/aer2006.pdf

U.S. Energy Information Administration (2009). *State energy consumption estimates 1960 through 2009* (DOE/EIA-0214). Washington, D.C.: Author. Retrieved from http://205.254.135.7/state/seds/sep_use/notes/use_print2010.pdf.

U.S. Environmental Protection Agency. (2003). *Estimating 2003 building-related construction and demolition materials amounts.* Washington, D.C.: Author. Retrieved from http://www.epa.gov/osw/conserve/rrr/imr/cdm/pubs/cd-meas.pdf

U.S. Environmental Protection Agency. (2011). *Inventory of U.S. greenhouse gas emissions and sinks: 1990-2009* (EPA 430-R-11-005). Washington, D.C.: Author. Retrieved from http://www.epa.gov/climatechange/Downloads/ghgemissions/US-GHG-Inventory-2011-Complete_Report.pdf.

U.S. Environmental Protection Agency. (2012). *Waste Reduction Model Version 12: Carpet*. Washington, D.C.: Author.

U.S. Environmental Protection Agency, Department of Transportation. (2011). *Final Rulemaking to Establish Greenhouse Gas Emissions Standards and Fuel Efficiency Standards for Medium- and Heavy-Duty Engines and Vehicles* (EPA-420-R-11-901). Washington, D.C.: Author.

U.S. Housing and Urban Development. (2001). *A report on the feasibility of deconstruction: An investigation of deconstruction activity in four cities*. Washington, D.C.: Partnership for Advancing Technology in Housing. Retrieved from http://www.huduser.org/publications/pdf/deconstruct.pdf

U.S. President's Council of Sustainable Development. (1999). *Toward a sustainable America*. Washington, D.C.: Author. Retrieved from http://clinton2.nara.gov/PCSD/Publications/tsa.pdf

Walls, M. (2006). *Extended producer responsibility and product design: Economic theory and selected case studies* (RFF DP 06-08). Washington, DC: Resources For The Future.

Walters, B. J., & Wheeler, J. O. (1984). Localization economies in the American carpet industry. *Geographical Review, 74*(2), 183-191.

Wang, Y., Zhang, Y., Polk, M., Kumar, S., & Muzzy, J. (2003). Recycling of carpet and textile fibers. In A. L. Andrady (Ed.), *Plastics and the environment,* Hoboken, N.J.: Wiley-Interscience.

Warner, M. E., & Bel, G. (2008). Competition or monopoly? Comparing privatization of local public services in the US and Spain. *Public Administration, 86*(3), 723-735.

Weber, C., Matthews, D., Venkatesh, A., Costello, C., & Matthews, H. S. (2009). *The 2002 US benchmark version of the economic input-output life cycle assessment (EIO-LCA) model*. Green Design Institute, Carnegie Mellon University. Retrieved from http://www.eiolca.net/docs/full-document-2002-042310.pdf

West, G. R. (1986) A stochastic analysis of an input-output model. *Econometrica*, 54, 363-374.

Williamson, O. E. (1985). *The economic institutions of capitalism: firms, markets, relational contracting*. New York, London: Free Press.

Williamson, O. E. (1989). Transaction cost economics. In R. Schmalensee & R. D. Willig (Eds.), *Handbook of industrial organization*. New York, N.Y., U.S.A.: North-Holland.

Wilburn, D., & Gooman, T. (1998). *Aggregates from natural and recycled sources: Economic assessments for construction applications-A materials flow analysis.* (U.S. Geological Survey Circular 1176). Denver, Colorado: U.S. Department of Interior.

Wood, G. A., & Parr, J. B. (2005). Transaction costs, agglomeration economies, and industrial location. *Growth & Change, 36*(1), 1-15.